Vegetation recovery following the 2015 Flock Hill fire, Canterbury high country

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Abstract

This study presents the results of the investigation into the vegetation recovery following the 2015 fire at Flock Hill Station.

In 2015 a large fire burnt an area of approximately 300 hectares at Flock Hill Station. This area was largely dominated by the invasive wilding conifer *Pinus contorta* and exotic pasture grasses such as *Agrostis capillaris* and *Anthoxanthum odoratum*. The fire also burnt areas of native shrubland, consisting largely of *Leptospermum scoparium* and *Discaria toumatou*, and mountain beech forest (*Fuscospora cliffortioides*).

The strongest post-fire recovery was shown by exotic pasture grasses and herbaceous weeds. The native vegetation showed very little recovery following the fire. *Discaria toumatou* was the only native woody species to show an ability to re-sprout following fire whilst *Viola cunninghamii* and *Wahlenbergia albomarginata* showed the highest levels of recovery of native herbaceous species.

Pinus contorta showed very little ability to recover following the fire. *Pinus contorta* showed very little germination after the fire and tests carried out on seeds extracted from burnt cones showed very low viability.

This research could have important implications for future wilding pine control and also for the management of native vegetation in the Canterbury high country as the risk of fire increases in this area with climate change.

Introduction

When a new species is introduced into an area it can have a large impact on the ecosystem into which it is introduced (McIntosh et al. 2012) and can threaten conservation and landscape values (Ledgard 2009). This is the case in New Zealand with introduced conifers. The term wilding conifers refer to exotic conifers (Pinophyta) that are able to regenerate naturally in the wild (Wardle 1985a, Ledgard 2009). These wilding conifers are especially prevalent in the degraded grasslands of the eastern South Island high country (Ledgard 2006, Ledgard& Paul 2008). Conifers were originally planted in the South Island high country in the late 1880s for wind and stock shelter, erosion control, timber and for firewood (Ledgard 1985, Ledgard 1989, Gous et al. 2015). As the number of farmed stock and wild animals decrease in high country areas due to land becoming more marginal and less economical, or through reversion to full Crown ownership and management, the opportunities for wilding spread are increasing (Ledgard 1997 – unpublished data). Wilding conifers now affect 0.6-0.8 million ha of South Island high country (Paul &Ledgard 2009), with more than 200,000 ha of this being on lands administered by the Department of Conservation (DOC) (Gous et al. 2014). Wilding conifers are a serious ecological and economic issue (Gous *et al.* 2015). Wilding conifers are a major ecological issue because they are able to outcompete native vegetation and modify ecosystems (Gous *et al.* 2015). The spread of wilding conifers is believed to have been occurring for over 100 years, but in recent years it has been receiving more attention (Ledgard 2008). DOC spend 25% of its annual $14 million weed budget on the control of wilding conifers (Ledgard 2009), and Environment Canterbury (ECan) spent 53% of its annual plant pest control budget on wilding control in 2008 (Ledgard 2009a). The main methods used in controlling wilding conifers are manually pulling them out of the ground, mechanical control (chainsaw) or chemical control through the use of herbicides (Gous *et al.* 2015). However, some are now asking the question as to whether fire can also be used to control wilding conifers.

Most pines are pioneer species; this means that their seeds germinate in open grassland and shrubland where there is high light and they can quickly dominate an area (Ledgard& Paul 2008). However, it is not just open grasslands and shrublands that can be invaded and displaced by wilding conifers. *Pseudotsuga menziesii* is more shade tolerant than species in the *Pinus* genus and is able to germinate and grow under a native forest canopy, mainly beech forests (*Fuscospora* & *Lophozonia*), although the numbers of *P. menziesii* in native forests is still low (Ledgard 2002). The shade tolerance of *P. menziesii* is one of the traits that make this species such a desirable forestry tree species in New Zealand, especially for continuous cover forestry. This shade tolerance trait combined with better timber durability, less maintenance and the excellent conditions for growing *P. menziesii* are helping to increase its popularity in New Zealand (Ledgard, Knowles & De La Mare 2005), especially in the South Island high country, where it is becoming the favoured plantation species (Ledgard 1997 – unpublished data). Since 1990 the area of *P. menziesii* plantations

has more than doubled with large plantation areas being established in the South Island high country (Ledgard *et al.* 2005), potentially increasing the risk of its spread.

Unlike *P. menziesii, Pinus contorta* is no longer planted in New Zealand (Ledgard 2001). This is due to low market acceptance because of poor tree form and wood quality, as well as the known ability for *P. contorta* to become invasive (Ledgard 2001). Despite *P. contorta* not being planted today it is still the most vigorously spreading wilding conifer in the South Island high country (Ledgard 2001, Ledgard& Paul 2008). This is in part a legacy of widespread plantings in the past (for shelter, timber production and erosion control), but also reflects the fact that *P. contorta* has the second lightest seed of all the members of the *Pinus* genus and therefore is capable of long distance wind dispersal (Ledgard 2001). *P. contorta* is a prolific seed producer and can start to produce viable seeds after just six years (Ledgard 2001). *P. contorta* also have aggressive fast-growing seedlings that are able to establish quickly, often out-competing the native vegetation in open grassland and shrubland (Ledgard 2006). *P. contorta* is the dominant conifer species at Flock Hill Station, which is where this research was carried out.

Flock Hill Station is a 14,400 ha high country sheep station in the Castle Hill Basin, Canterbury. In late January 2015, there was a major fire at Flock Hill Station which burnt around 300 ha of predominantly wilding *P. contorta* stands as well as native shrubland, some mountain beech forest and grassland. *P. contorta* is the dominant conifer species at Flock Hill Station forming almost pure stands in places, with a minor component of *P. menziesii* mixed through some of the stands. There is also scattered *Pinus mugo* and *Pinus silvestris* across the site, although these species are never common (Nick Ledgard pers. comm. 21 July 2015). The native woody vegetation in this part of Flock Hill Station is comprised shrubland comprising of *Leptospermum scoparium*, *Discaria toumatou*, *Hebe* spp and *Ozothamnus leptophyllus,* and mountain beech (*Fuscospora cliffortioides*) forest. Although fire does a good job of clearing a site of wildings, it can act as a double-edged sword in the sense that it also creates perfect conditions for the re-establishment of wilding conifers back into the site, whilst removing the native vegetation.

Pinus contorta is considered to be one of the most invasive species in the world and is a major pest plant species in Argentina, Australia, Ireland, New Zealand, Sweden, the United Kingdom, and Chile (Cobar-Carranza 2014).In Sweden there is a lot of concern around the use of *P. contorta* as a plantation species (Knight *et al.* 2001, McIntosh *et al.* 2012). In Northern and central Sweden *P. contorta* has been widely planted as a timber species (Hagner 1983) and there are concerns that due to its ability to rapidly colonise an area recently disturbed by fire that it will influence successional processes in Sweden's boreal forests and outcompete the native conifers following fire (McIntosh *et al.* 2012). Likewise in Chile, it has been shown that *P. contorta* increases the flammability of an area and that disturbance events such as wildfires helps *P. contorta* to outcompete the native vegetation and become the dominant species in an area (Cobar-Carranza 2014). Similarly, in New

Zealand there are concerns that following fire in areas colonised by wildings (such as the Flock Hill Station fire in early 2015), *Pinus contorta* will quickly colonise the burnt areas, preventing native vegetation from re-establishing in areas where it was present before the fire. This is because very few of New Zealand's native plant species are adapted to fire (Perry et al. 2014). In fact mānuka (*Leptospermum scoparium*) is the only native New Zealand plant that has traits suggesting adaptation to fire (Perry *et al.* 2014). These traits include the release of seeds during a fire (serotiny), woody valves fruit and seeds that are able to withstand moderate heat shock. Other species such as kānuka (*Kunzea* spp), bracken (*Pteridium esculentum*) and matagouri (*Discaria toumatou*) are pioneering species and are adapted to colonising open areas following a disturbance such as fire (Perry *et al.* 2014). However, this is often in natural areas where there are not high levels of competition from exotics. The situation is likely to be different when exotics (especially wildings) are dominant in and around a burnt area. The native plants that have fire adaptations or are able to rapidly colonise an area following a fire, such as mānuka or kānuka are members of the Myrtaceae family, which is Australian in origin, where wild fires are very common (Perry *et al.* 2014). Perry *et al.* (2014) also raises concerns around fire-adapted exotic species being favoured over New Zealand's native vegetation in current fire regimes but also in the future where climate change may result in more frequent fire events. The lack of fire adaptation in New Zealand's native flora is likely due to a lack of natural fires through New Zealand's pre-human history (Perry *et al.* 2014).

As there is real concern that the area burnt during the fire at Flock Hill Station will revert to wildings, it is important assess the vegetation recovery immediately following the fire to establish the response of wildings post-fire. Possible implications of this research are likely to be around the potential impacts of future climate change on the spread and increase in wilding conifers after fire, as well as the ability for native vegetation to respond to fire. Under future climate change scenarios, the prediction for the Canterbury high country (where wilding conifers are prolific in places) is that it is likely to get drier and windier (Perry *et al.* 2014). This is likely to increase the risks of more frequent and severe fires in the Canterbury high country, which may either hinder or assist the spread of wilding conifers. Increasing areas of the Canterbury high country are regenerating towards a native woody vegetation cover and fire could jeopardise this (Young et al. 2016), especially if wilding conifers are either present or have the potential to spread into the area after fire.

Wilding conifer spread

Up until the 1980s there was little concern about wilding conifers. In fact conifers in areas where wildings now occur were considered positively, being useful for erosion control, timber, firewood and shelter for stock (Ledgard& Belton 1985). Ledgard& Belton (1985) were the first to consider the invasiveness of five exotic conifer species present in the South Island high country; these were *Pinus nigra, P. ponderosa, P. radiata, Larix decidua* and *Pseudotsuga menziesii*. They found that *L. decidua* was the most vigorous in its spread with 62% of the studied stands producing wilding spread of seedlings (Ledgard& Belton 1985).

The second highest seedling spread came from *P. nigra* with 42% of stands having wilding spread, followed by *P. ponderosa* with 37%, *P. menziesii* with 36% and *P. radiata* the lowest with 25% of stands producing wilding spread (Ledgard& Belton 1985). They did not consider *P. contorta* in this study.

Over the last thirty years, a number of papers have been published investigating the spread of wilding conifers. Several factors have been identified that aid the invasion of wilding conifers. These include seed size, seed production the age at which cone production starts, the shade tolerance of certain species, altitudinal variation, and traits that make them successful pioneers (Wardle 1985a, 1985b, Ledgard 2001, 2002, Ledgard& Paul 2008). Ledgard (2001) found that of the invasive exotic conifers found in New Zealand, *Pinus contorta* is the most vigorous spreading. One of the reasons for this is due to *Pinus contorta* having such a light seed; in fact it has the second lightest seed in the *Pinus* genus (Ledgard 2001). *P. contorta* produces 200,000 to 300,000 seeds per kg and seeds can be carried a long distance in the wind, making wind a major contributor to wilding spread (Ledgard 2001). Winds from the NW are especially important and this potentially will be an increasing issue with climate change. The age of seed production is another factor that allows *P. contorta* to spread so vigorously (Ledgard 2001). *P. contorta* is capable of producing cones after just six years. The majority of the species in the Pinaceae family are pioneering species and are well equipped to establishing in open grasslands (Ledgard& Paul 2008). Due to their ability to establish in open areas wilding conifers have a high weed potential and their spread is often detrimental to the native flora and fauna and is especially invasive in the South Island high country (Ledgard 2001). As stands of wilding conifers age, the number of native species present drops dramatically as the canopy closes and native species tend to be absent. This is likely because the conifers invade open grasslands where the native species are not able to grow under a canopy (Ledgard& Paul 2008).

Invasions into native vegetation

Pseudotsuga menziesii is able to invade under a canopy due to being more shade tolerant than the other wilding conifers in New Zealand (Ledgard 2002). Ledgard (1989) and Ledgard (1996 – unpublished data) found that conifers readily spread throughout various forms of native vegetation in the Craigieburn Range. Ledgard (2002) found evidence of *P. menziesii* growing in native beech forests however these were low in numbers and none grew above five centimetres tall. However, in recently disturbed areas, such as following a fire or wind throw and on the edges of beech forests *P. menziesii* appears able to more readily colonise the area (Ledgard 2002). Ledgard (2002) found that at a site where *P. menziesii* had been actively planted under a mature beech canopy the survival rate for *P. menziesii* was around 50%, however the majority of the surviving *P. menziesii* looked unhealthy and only the plants in well-lit light-wells were able to grow above the beech canopy and start to produce cones. This shows that *P. menziesii* is likely only able to invade into a native beech forest where there has been a disturbance resulting in a relatively well lit area. However *P. menziesii* is able to invade into mānuka and kānuka shrubland (Ledgard 2002, Davis *et al.*

2011). Even under a closed canopy of mānuka and kānuka there is still sufficient light for *P. menziesii* to establish (Ledgard 2002, Davis *et al.* 2011).

All the exotic wilding conifers in New Zealand are able to establish in open native grasslands and native scrublands but *P. contorta, P. nigra, P. mugo, P. ponderosa* and *P. menziesii* are the most invasive (Ledgard& Belton 1985, Ledgard 2001, Gous *et al*. 2010). This is due to their being pioneering species back in their native ranges, where they colonise open areas following disturbances (Ledgard 2001). Wardle (1985a) found that *P. contorta* is able to establish itself well above the native treeline. In an experiment carried out in the Craigieburn Range, Wardle (1985a) investigated the ability of both native and exotic tree species to germinate and grow at different altitudes and found that *P. contorta* was able to germinate and grow at all altitudes measured. At the highest tested altitude (1750m), which was well above the natural treeline (c. 1400 m), it was the only tested species that was able to grow in the native alpine flora. Although *P. contorta* is not a treeline species in its home range of North America, it is able to establish well above the New Zealand treeline even when the other treeline species from its native home range cannot (Wardle 1985a).

Control options for wilding spread

There are several ways that wilding conifers have been controlled over the years, each with differing success. These include herbicide treatments, fertilising and grazing, and various removal methods such as harvesting or felling to waste (Ledgard& Norton 2008, Paul &Ledgard 2009, Gous *et al.* 2010). Gous *et al.* (2010) looked at the effects of different herbicides on *Pinus contorta*, *Pinus mugo* and *Pseudotsuga menziesii.* They found that Triclopyr-based herbicides are more effective that glyphosate-based herbicides (Gous *et al.* 2010). This was backed up by later experiments which looked at aerially applied herbicides for wilding control and at whether aerially applied tryclopyr-based herbicides can be an effective way of controlling dense stands of *Pinus contorta* Gous *et al* 2014, 2015). When applying herbicides, spray volumes and droplet size are very important (Gous *et al.* 2014). Coarse droplet size applied at low spray volumes is ineffective for controlling wilding conifers because it offers poor foliage cover (Gous *et al.* 2014). However, when applied at a high spray volume with a very large droplet size aerially applied tryclopyr can be an effective control of *P. contorta* (Gous *et al.* 2015).

Fertiliser application to degraded grasslands of the South Island high country has been investigated as a control option for wilding conifers (Ledgard 2006). It has been thought that fertilising degraded grasslands would reduce the numbers of wilding conifers through increased competition from other plants whose growth would be enhanced by the fertiliser (Ledgard 2006). Ledgard (2006) found that through the use of fertiliser application, the number of wilding conifers was reduced, in some places by up to an average of 42%. However in areas where wilding conifers grow there will always be need for wilding control. Fertiliser application could also allow for more grazing which is thought to be another way in which wilding spread could be controlled (Ledgard 2006, Ledgard& Norton 2008). Sheep

grazing can greatly reduce the establishment of wilding conifers and through the use of fertiliser, stock grazing can be increased allowing for a reduction in the spread of wilding conifers. Grazing of wilding conifers by rabbits can also reduce the establishment of wilding conifers (Ledgard& Norton 2008). However, Ledgard& Norton (2008) raise concerns about using fertiliser and stock grazing as control methods for wilding control as they are also likely to prevent the establishment of native vegetation.

Various removal methods have been used to control wilding conifers including felling, harvesting and poisoning (Paul &Ledgard 2009). When wilding conifers are felled to waste, native grass and shrub species tended to grow better in the area of dead trees, possibly because the increase in nutrients from the decaying trees and creation of a novel microsite and shelter helped to give an advantage to the native species (Paul &Ledgard 2008). This is in contrast to the open grassland vegetation found in the South Island high country which tends to be dominated by exotics such as *Pilosella* species (Paul &Ledgard 2008). However, when Paul &Ledgard (2009) compared vegetation compositions following different wilding control methods; including felling, mulching, harvesting and poisoning they found that felling resulted in rapid plant growth but that these affects were not long-lasting and over time there was a reduction in biodiversity. Paul &Ledgard (2009) also found that mulching initially reduced plant growth but offered little wilding control in the long term. Harvesting helped the establishment of exotic species including wilding conifers, however encouragingly the only vegetation found growing under the poisoned trees were native vegetation.

There have been guidelines produced for foresters and land managers to help them reduce the risks of their plantations producing wilding spread (Ledgard& Langer 1999, Ledgard 2004, Ledgard 2008). These include scorecards that allow foresters and land managers to calculate the risk of wilding spread (Ledgard& Langer 1999, Ledgard 2008). These have focused on planted species, planting site and the nature of down-wind land management.

Potential for an increase in the spread of wilding conifers

The increasing popularity of *P. menziesii* as a forest timber species has potential to result in increased wilding spread (Ledgard *et al.* 2005). *P. menziesii* has a number of advantages over *P. radiata* such as better timber durability and quality, better market value and require no pruning*;* however, one of the big disadvantages is the tendency for *P. menziesii* to have wilding spread (Ledgard *et al.* 2005). Maclaren (2005) released a paper looking at realistic alternatives to *P. radiata* for forestry in New Zealand. In this paper it is suggested that *Pinus nigra* could be a good alternative to *P. radiata* in inland Canterbury where *P. radiata* struggles to grow; however as *P. nigra* is a major pest plant in this region (Ledgard& Langer 2004) this has the potential to increase the wilding problem.

Effects of fire on pines

The *Pinus* genus in widely considered to be a fire adapted genus (Tryterud 2003). Ways in which pines can show adaptations to fire include having thick bark, the ability to self-prune, the ability to re-sprout, serotonous cones, and germination that is triggered by smoke, ashes and heat-shock (Cobar-Carranza 2014). Large areas of the Northern Hemisphere are covered in forests that are dominated by pines and fire has been a common disturbance within these areas (Wooster & Zhang 2004, Ohlson*et al*. 2011). The coniferous forests of Europe and North America are important in the uptake of anthropogenic carbon (Kashian*et al*. 2013) and because of this there have been a number of studies looking into the effects of fire on pines (Kuosmanen 2014). This is especially true for the conifer dominated boreal forests where fire has traditionally been the major disturbance agent and ecological factor controlling the vegetation structure (Lehtonen & Kolstrom 2000, Groven & Niklasson 2005, Ivanova *et al*. 2010). Studies carried out in the European boreal forests looking at the historic effects that fire has had on the region have found that frequent fires during the Holocene period resulted in forests that were dominated by pines (Lindbladh, Niklasson & Nilsson 2003, Greisman & Gaillard 2009, Brown & Giesecke 2014) and in areas with less frequent fires were dominated by spruce (*Picea* species) and larch (*Larix* species) species (Pitkanen *et al*. 2002, Kousmanen *et al*. 2014). Since the 18th century slash and burn agriculture has become a common practice in Fennocandia (Groven & Niklasson 2005) causing large scale disturbances in the landscape and resulted in large areas regenerating in pine (Engelmark, Kullman & Bergeron 1994, Engelmark, Hofgaard & Arnborg 1998). It is partly due to this response to fire shown by pine that fire has now become the major tool for forest restoration in Fennoscandia (Vanha-Majamaa*et al*. 2007, Eriksson *et al*. 2013).

Like in Europe, fire is a major agent of disturbance in North American coniferous forests (Smithwick*et al*. 2005) and like in Europe, the North American pine species also show an adaption to fire (Smithwick*et al*. 2009). *Pinus contorta* is one North American species that shows a high level of fire adaptation (Alexander 1980, Smithwick *et al*. 2005, Smithwick *et al*. 2009).It is believed that frequent fires in North America during the mid-Holocene aided the migration of *P. contorta* north into the Yukon Territory, Canada (Strong & Hills 2013, Edwards *et al*. 2015). The current expansion of *P. contorta* in north-west Canada has also been linked to fire (Edwards *et al*. 2015). Fire appears to act as a trigger for population expansion for *P. contorta*, which ranges from Yukon Territory, Canada, in the North to California, United States, in the south (Pierce & Taylor 2011, Edwards *et al* 2015). *Pinus contorta* shows high levels of serotony in its native range with high levels of serotonous cones produced in stands that have recovered following a fire (Muir & Lotan 1985, Turner *et al*. 2007). This has even been found to occur in very young stands of approximately fifteen years of age (Turner *et al*. 2007). *Pinus contorta* has also been found to alter the fire regime in ecosystems outside of its native range after invading (Cobar-Carranza 2014). Cobar-Carranza (2014) found that after *P. contorta* had invaded into the Malalcahuello Reserve in Chile that the flammability of the forest increased due to the change in fuel type, specifically

and increase in the highly flammable *P. contorta* fuel (Cobar-Carranza 2014). *Pinus* species have been found to invade into areas affected by fire in South Africa and there are fears that with the potential for more frequent and severe fires in the Malalcahuello Reserve in Chile that *P. contorta* will increase its range with fire favouring *P. contorta* regeneration (Cobar-Carranza 2014).

Effects of fire on native vegetation

Prior to human arrival, fire was uncommon in New Zealand (Perry *et al.* 2014). However, following human arrival, Polynesian fires have had a large impact on New Zealand's native vegetation (Wardle 2001, Ogden *et al.* 2003, Williams 2009). This is because New Zealand has vegetation that lacks significant fire adaption or serotiny (Perry *et al.* 2014). Upon their arrival in New Zealand, Early Polynesians began to regularly burn the existing vegetation for agriculture and to clear travel routes. Perry *et al.* (2014) suggest that mānuka is the only true fire adapted species in New Zealand and that other species such as kānuka, bracken and matagouri are adapted to colonising an area after a disturbance but are not adapted to fire. Most of New Zealand's fire adapted and rapid coloniser flora are closely related to Australian families such as Dennstaedtiaceae, Myrtaceae and Rhamnaceae (Perry *et al.* 2014). Most of these species are relatively new (early Miocene) arrivals to New Zealand (Perry *et al.* 2014). Some believe that the serotiny of mānuka has allowed it to become common throughout the South Island where there has been a history of fires following human arrival (Bond et al. 2004).

Most of New Zealand's native species do not have any fire adaptations and take a long time to re-colonise an area following a fire. For example, beech can take decades, even centuries to re-establish in to an area affected by fire (Wiser et al. 1997, Allen 2001). Perry *et al.* (2014) raised concerns that while New Zealand has a lack of fire adapted natives, there is a suite of exotic vegetation in New Zealand that are highly adapted to fire including species from the *Pinus, Ulex* and *Acacia* genera. They express a concern that with climate change may come more frequent fires and it may be difficult for the native vegetation to compete with the fire adapted exotic species.

Aims

This study aimed to investigate the impacts that fire has on the vegetation of a high country ecosystem. The overall recovery of the vegetation post-fire was investigated with special interest placed on the recovery of both the native vegetation and the recovery of *Pinus contorta*.

Hypotheses

1) As the New Zealand flora evolved largely in the absence of fire (Perry *et al.* 2014), native vegetation is unlikely to be able to recover after fire.

2) *Pinus contorta* will dominate burnt vegetation, both native and exotic, because of its serotinous cones and the ready dispersal of its light seeds into burnt sites.

The two hypotheses were tested by investigating the ability of all burnt woody species in the established transects to re-sprout and by investigating the ability of all vegetation types to re-establish following the fire in the established transects.

Study site

Location

The study site was located at Flock Hill Station, a 12,000 ha high country farm in the Waimakariri Basin, Canterbury (43°09'30.79" S 171°45'15.43" E). Flock Hill Station is approximately 100km west of Christchurch and is located along the State Highway 73. The study site is located in the Cass Ecological District.

The topography of the burnt site at Flock Hill Station is steep to undulating in most parts with flatter terraces present across the site (see figure 1). There is a large stream that runs through Dannys Gully, with erosion channels leading down into it from the steeper scree slopes above (Taylor 2015). All of the study plots are located between 900m and 1100m.

Figure 1. A view of the study site from Helicopter Hill.

Geology and Soils

The study site is a remnant of a tertiary aged bed of marine sediments with mudstone and sandstone covered by limestone. Weathered limestone tors are also present at Flock Hill Station (Shanks *et al.* 1990, Cox &Barrell 2007). The soils are comprised of sand and clay and overlie Mesozoic greywacke (Pole &Vajda 2009, Relph 1957).

Climate

The climate is usually warm in the summer and cool in the winter which is typical for an intermontane basin (Shanks *et al.* 1990). The predominant wind direction is from the west (McCracken 1980). Under these westerly conditions Flock Hill station is often dry and mild, however when troughs from the Tasman Sea approach, they result in lowering air pressure and strong north-west winds creating warm, dry conditions. Under very strong north-west winds it is not uncommon for precipitation from the west to reach Flock Hill Station (McCracken 1980). However, the majority of the precipitation comes from the south-west. These south-west winds often result in cooler temperatures and increased levels of precipitation and, in winter, snow, which can persist on the ground for several weeks or months (McCracken 1980, Shanks *et al*. 1990). The annual precipitation at Flock Hill Station is between 900 to 1200mm with an average annual mean temperature of 8^0C, with average monthly temperatures ranging from 14.6^0C in January down to 1.6^0C in July and winter temperatures dropping as low as -10^0C (McCracken 1980, Shanks *et al*. 1990). Rainfall is at its highest during spring with the driest period in February and March (Shanks *et al*. 1990).

Historical vegetation

The historical vegetation cover at Flock Hill Station was dominated by beech forest prior to human arrival in New Zealand (Paul &Ledgard 2006, Young, Norton & Lambert 2016). However, since the first human arrivals in the 13^{th} century AD and following European colonisation in the 17^{th} century AD the vegetation structure has changed significantly (McGlone & Basher 1995, McWethy *et al*. 2009, Young *et al.* 2016). Fire was the major cause of this change with fire being used for land clearance (Paul &Ledgard 2006, Knight *et al*. 2009, Young *et al*. 2016). This frequent use of fire resulted in the loss of mountain beech from large areas and a shift to native shrubland and grasses. European pastoralism resulted in the introduction of exotic pasture grasses, grazing and further burning. This resulted in a shift to pasture grasses being the dominant vegetation type with scattered shrubland species and small pockets of beech forest (often only in the gullies where it is hard to burn) (Paul &Ledgard 2006, Young *et al.* 2016).

Current vegetation

The current vegetation present at Flock Hill Station consists mainly of exotic pasture grasses and herbs such as browntop (*Agrostis capillaris*), sweet vernal (*Anthoxanthum odoratum*), white clover (*Trifolium repens*) and chewings fescue (*Festuca rubra*). There are also native grasses present such as *Chionochloa* species, *Poa* species and *Festuca* species.

Also present are areas of native shrubland which consist largely of matagouri (*Discaria toumatou*), mānuka (*Leptospermum scoparium*), *Dracophyllum acerosum*, *Ozothamnus leptophyllus*, *Coprosma propinqua*, *Aristotelia fruticosa* and *Pimelia* species. Native mountain beech (*Fuscospora cliffortioides*) forest is also present in some of the gullies at Flock Hill Station. A number of exotic weed species are also present including European broom (*Cytisus scoparius*), hawkweeds (*Pilosella officinarum*, *Pilosella praealtum* and *Hieracium lepidulum*), sweet briar (*Rosa rubiginosa*), sheep's sorrel (*Rumex acetosella*) and wilding conifers, including *Pinus contorta*, *Pinus mugo*, *Pinus sylvestris* and *Pseudotsuga menziesii*. Of these exotic species *P. contorta* is the most widespread. Wilding *P. contorta* now covers an extensive area of Flock Hill Station.

History of pine invasion at Flock Hill Station

During the 1940s and 1950s the National Forest Survey raised serious concerns about vegetation loss and erosion in the high country which was believed to be due to excessive fires and grazing from both stock animals and wild animals. In 1956 the New Zealand government established a forest and range experiment station in Rangiora, North Canterbury. This research station was part of the Forest Research Institute. The re-vegetation division of this station was based in the Craigieburn Forest Park. Close to 400 species were planted in the Craigieburn area as part of an erosion control research trial, of the nearly 400 species, 90 were exotic conifers. One of the exotic conifer species trialled was *Pinus contorta*, which grew very well in the Craigieburn area. However, during the 1960s it was recognised that there was a high risk of natural regeneration associated with *P. contorta* and its use was discontinued at Craigieburn. By this stage however, the damage was done and *P. contorta* quickly established and began to spread throughout the Craigieburn Forest Park and then into Flock Hill Station which was down-wind. The first *P. contorta* at Flock Hill Station were observed in the 1970s and since then, trees have spread from outlier trees scattered across the site, with most of the trees being the progeny of the original colonisers. A major increase in the spread of *P. contorta* occurred in 2002 due to a change in the leasehold that year resulting in a large reduction in sheep numbers on the station. This resulted in a large reduction in stock grazing, allowing large numbers *P. contorta* seedlings to establish (see figures 2 and 3). Wilding conifers now cover a large area of Flock Hill Station with 95% of these being *P. contorta*. The other 5% are made of *Pinus mugo*, *Pinus sylvestris*, *Pinus ponderosa*, *Pseudotsuga menziesii* and *Larix decidua*.

Figure 2. Flock Hill Station 2001 prior to leasehold change. Photo courtesy of N Ledgard.

Figure 3. Flock Hill Station 2007 after leaseholder change and low grazing. Photo courtesy of N Ledgard.

Fire

The Flock Hill fire started on the 26th of January 2015 at approximately 2:30pm. The fire originated on the eastern side of State Highway 73 at the Craigieburn cutting with the likely cause of the fire being a tractor seen travelling up this road shortly before the fire started.

One of the major factors helping the fire spread were three wind direction shifts on the first day. At the time the fire started the wind direction was North West, this helped to push the fire to the South East allowing it to burn up onto the top terrace and allowing the fire to jump across Danny's Gully at the hairpin corner of the Craigieburn cutting and spread above the farm track. At approximately 6:30pm the wind direction changed to the South West. The third and last wind direction change occurred at approximately 8:30pm, this pushed the fire to the South West allowing it to burn down slope and formed a smoke column that filled the Castle Hill basin and crated fears of spot fires starting at Cave Stream and Castle Hill village from embers falling out of the smoke column. At the time of the fire the weather conditions were warm and dry with a temperature of 25.12 ^{0}c and a relative humidity of 19.75% (Taylor 2015 – unpublished data).

The major fuel types for the fire was wilding pines, grasses, beech trees and native shrubs. The dominant fuel type across the fire area was wilding pines. These wilding pines grew in dense stands that covered large areas, especially on the top terraces and on the slopes down towards Cave Stream. The wilding pines ranged in size from 2m tall to 10m+ in height. As well as the living dense stands of wildings there were also areas of standing dead trees that had been sprayed in previous years. The main fuel on the upper slopes was the native shrubs consisting of mānuka, matagouri, *Hebe*, *Coprosma* and *Dracophyllum* shrubs. Grasses were another key fuel. Tests were carried out by Scion measuring the moisture content of the grass around the area of the fire origin. The results showed the grass to have a moisture content of 13%. Beech was another key fuel type. An area of tall beech trees was present around Danny's Gully. Under the beech canopy was a large amount of decaying wood which was sitting on a thick layer of duff. The areas of burning beech took most of the fire suppression resources over the several days the fire was burning (Taylor 2015 – unpublished data).

The fire's behaviour was classed at the extreme end with the fire spreading rapidly both across the terrain and via wind. Long range spotting was common on the first afternoon with spotting often occurring 250m away and in some cases up to 500m away. The fire burnt between 720m and 1240m elevation with the fire spreading rapidly up the steep faces of Broken Hill and Jamies ridge with the fire becoming well established and long burning in the wilding pines and beech forest, with flame length being estimated as close to 30m in places (Taylor 2015 – unpublished data).

The cause of the fire is believed to be from an old tractor observed driving uphill past the fire site under a heavy load and ejecting hot carbon from the exhaust. This resulted in thirteen points of origin on the uphill (east) side of SH 73 with a fourteenth point on the other side of the road after the tractor turned around and headed back down hill. The fire burnt for around four days and burnt approximately 300 hectares of vegetation (Campbell 2015 – unpublished data).

Methods

Vegetation sampling

To investigate the response of the vegetation following the January 2015 fire, forty 20x2m transects were established amongst different pre-fire vegetation types and were sampled throughout the first summer following the fire and once again in the second spring following the fire. The transects were permanently marked and were used to investigate the vegetation recovery. The main pre-fire vegetation types were identified and used as a basis for sampling. Five vegetation types appear to have been present prior to fire (see figure 9):

(1) Mixed native shrubland containing young, newly established *P. contorta*(mixed shrubland with young *P. contorta*)(see figure 4);
(2) Mixed native shrubland containing older, well established *P. contorta* (mixed shrubland with old *P. contorta*)(see figure 5);
(3) Dense stands of *P. contorta* which had been sprayed prior to the fire (sprayed *P. contorta*)(see figure 6);
(4) Dense stands of *P. contorta* that had not been sprayed (dense P. contorta)(see figure 7);
(5) Native mountain beech forest (beech) (see figure 8).

Figure 4. Mixed shrubland with young *P. contorta* vegetation type

Figure 5. Mixed shrubland with old *P. contorta* vegetation type

Figure 6. Sprayed *P. contorta* vegetation type

Figure 7. Dense *P. contorta* vegetation type

Figure 8. Beech vegetation type

Vegetation recovery plots

Forty 20x2 m belt transects were established (Figure 10(map)), eight transects in each of the five vegetation types. Each transect was permanently marked with metal pegs in each of the four corners and flagging tape for easy finding. When measuring, each transect was marked out using a measuring tape and then divided up into five 2x2m quadrats. Transects always ran across the slope (i.e. along the contour line) and were always located completely within one vegetation type (see figure 11).

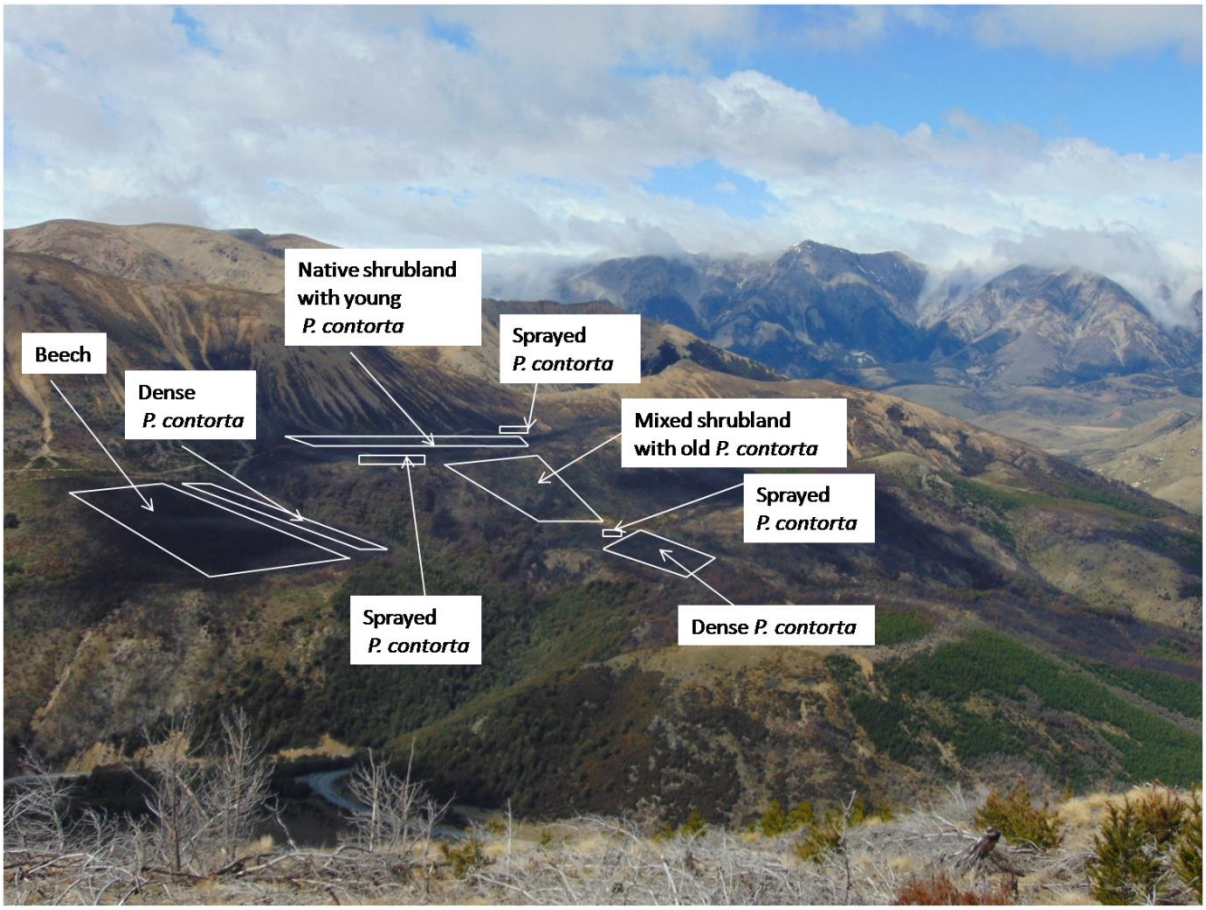

Figure 9. Location of vegetation types within the burnt area

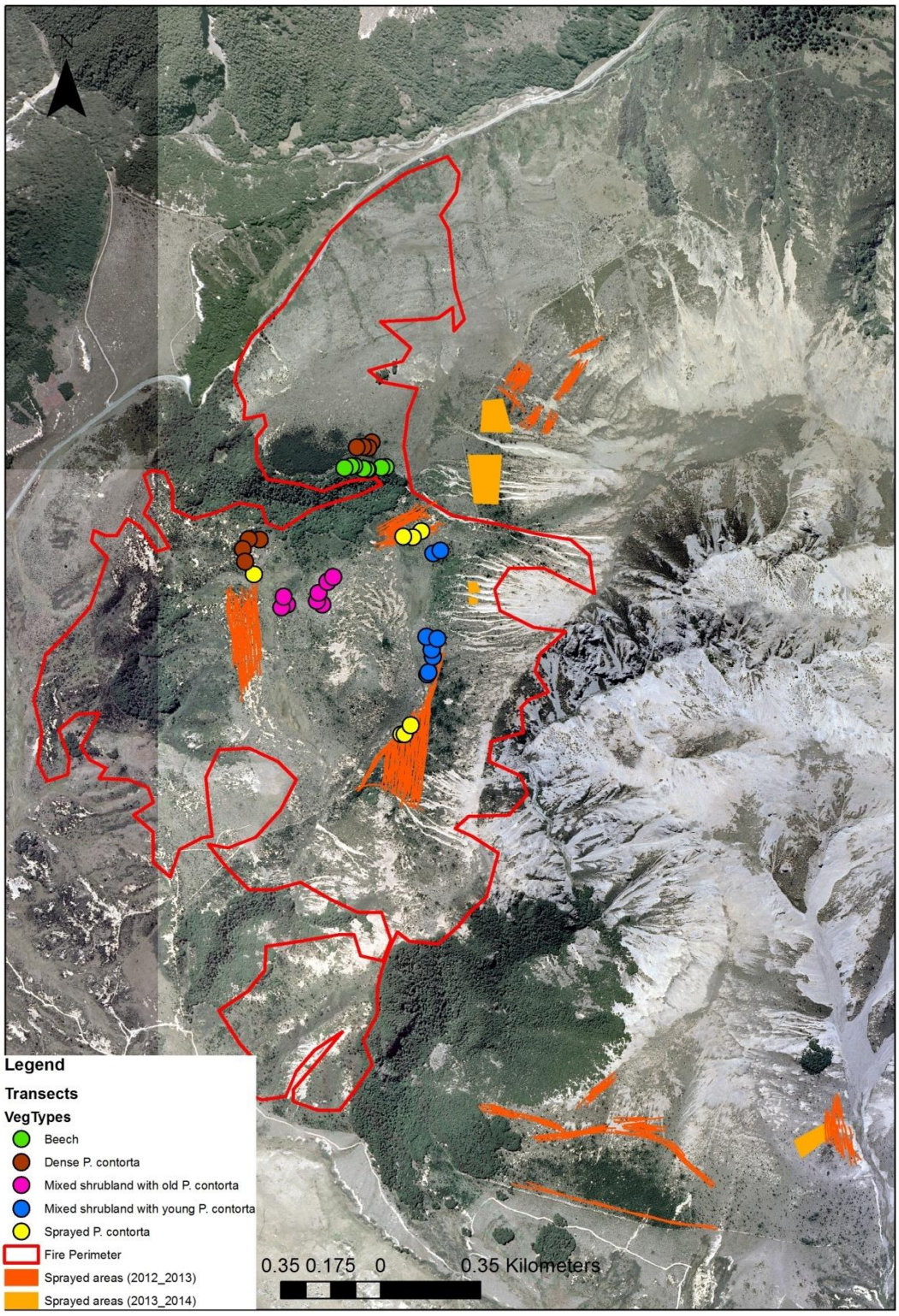

Figure 10. Map showing the area burnt, the areas of sprayed wilding *P. contorta* and the location of the 40 transects. The entire burnt area was over sown with *Lolium perenne*, *Dactylis glomerata*, and *Trifolium repens* following the fire.

Figure 11. laid out transect in the Mixed shrubland with young *P. contorta* vegetation type.

Five 2 x 2 m quadrats were located at 0–2m, 4–6m, 9–11m, 14–16m, and 18–20m along the 20m transect and used for measurements. Within these quadrats species presence and cover abundance was recorded for each vascular plant species present and also bryophytes as a single group. The cover abundance of each species was recorded as a score of 1 – 6 with each score correlating to a percentage range of cover (1 = less than 1% cover, 2 = 1–5% cover, 3 = 6–25% cover, 4 = 26–50% cover, 5 = 51–75% cover, and 6 = 76–100% cover). The total cover percentage of all plants was also recorded in each quadrat as well as the total percentage cover for dead material and bare ground. This was estimated to the nearest percent. The presence of animal droppings was recorded to gain an idea of what animals were visiting the transects in order to understand what animals might be grazing the transects.

All forty transects were measured three times over the 2015 – 2016 summer. Once in early December 2015, once in early February 2016, and once in early April 2016 to monitor the change in vegetation over the course of the summer and again in early November 2016 to monitor any changes that occurred over winter and in the early part of the second growing season following the fire.

A general linear model test was carried out on the data using the statistical computer program R to assess whether time or pre-fire vegetation type had an effect on the recovery

of the vegetation within the transects. Each species score for the five quadrats was averaged out to provide a single score for each species in each transect for all four months. The average scores for each species were then given a percentage cover that related to the average score, which was the midpoint of the percentage range for that score. A species with a score of ≤1 was given a percentage of 0.5, a score of 1.1-2 = 3%, 2.1- 3 = 17.5%, 3.1 - 4 = 37.5%, 4.1 – 5 = 62.5%, 5.1 – 6 = 87.5% (however no species averaged above 5). The percentages were then added together to get the average percentage cover for native and exotic species and also for all grass species, all herbaceous species, all woody species, and all bryophyte species. A general linear model test was also used to assess whether time and the pre-fire vegetation type had any influence on the number of species in a transect. The number of species in a transect was established simply by counting the number of species in each transect on the dataset. The species were split into native and exotic species as well as all grass species, all herbaceous species, and all woody species. The bryophytes were not analysed for this test as no one involved in the study had the ability to identify different moss species, so all moss species were grouped under the title moss and were only used to establish bryophyte cover. An ordination was also carried out on the data using the 'vegan' package in the statistical programme R. for the ordination data each species score for the five quadrats was averaged out to provide a single score for each species in each transect for all four months, Just like in the percentage vegetation cover analysis, however no further alterations were carried out on the dataset.

Herbivore exclusion cages

Two 50x50cm herbivore exclusion cages were established in each of the forty transects to investigate the effects of herbivory on the recovering vegetation. Unfortunately the majority of the herbivore exclusion plots were damaged by unexpected livestock (cattle) which meant livestock and other herbivores were able to graze freely on the post-fire vegetation in these plots. Because of this it was not possible to assess the true effect that grazing is having on the post-fire vegetation recovery.

Photo monitoring

Photo monitoring was used as a visual guide to show how the burnt areas were changing over time. Photo monitoring was carried out on the vegetation recovery transects, a wooden stake was placed at each end of the transect to permanently mark the point in which the photo should be taken from. The wooden stakes were placed 2.5m from either end of the vegetation recovery transect and photos were taken at the establishment of the plots and also at each measurement of the plots. Photos were taken looking down the transect from both ends.

Re-sprouting monitoring

The re-sprouting abilities of all woody plants was assessed in 25x4 m transects centred on the 20x2 m belt transects used for vegetation monitoring. The re-sprouting transects ran from one photo monitoring stake to the other and extended 1 m either side of the vegetation monitoring transect. These transects were marked out using a measuring tape and the species of burnt woody vegetation were identified; the main way in which the burnt species were identified were by their growth form and by their bark. The height of the burnt woody vegetation was measured and scored into one of four categories: 1) below 10cm in height, 2) between 10cm and 30cm in height, 3) between 30cm and 1m in height, and 4) over 1m in height. The burnt woody vegetation was then checked for any signs of re-sprouting. Any re-sprouting plants were recorded by species and height, and flagging taped for re-check later in the summer. The height of all plants was recorded to help investigate whether the size of the plant had any influence on re-sprouting ability. At the end of the summer (April) the re-sprouting plants were re-checked to investigate if the re-sprouts had survived to the end of the summer. All surviving re-sprouts were again measured and recorded to enable the investigation of how many re-sprouts survived through the summer and whether size had an influence in this. The re-sprouts were checked again in November 2016 to see if the re-sprouts survived over the winter. An analysis of variance test was carried out on the data to assess whether time or the size of the plant had an effect on the re-sprouting of *Discaria toumatou* within the transects. Sample sizes were too small for analysing data for other species.

A two way anova test was carried out on the data using the statistical computer program R to assess whether time or plant size had an effect on the re-sprouting ability of *Discaria toumatou*. The percentage of re-sprouts for each plant size in each month was established to allow for the statistical analysis.

Mānuka sowing

Mānuka (*Leptospermum scoparium*) seeds were sown at one end of each of the forty transects to investigate whether mānuka is able to establish post fire and whether sowing mānuka after a fire is a viable tool for restoration. At the northern end of each transect a palm full of mānuka was spread over a 15x15cm area of disturbed soil. Loose soil was then placed over the area to cover the seeds and prevent them from desiccation and being blown away in the strong northwest winds. The mānuka seeds were sown in late November 2015. The mānuka seeds were collected from an un-burnt area of mānuka located at Flock Hill Station. The sites where the mānuka was sown were checked in April 2016 and again in November 2016 to see if any mānuka germinated.

Seed viability testing

To test the viability of *P. contorta* seeds after the fire, cones were collected from trees that had been burnt in the fire. Ten cones were picked from each of ten trees for each of two burn classes. The first burn class was lightly burnt cones which was defined as cones that had only suffered light burning with no structural damage having occurred to the cones scales. The second burn class was heavily burnt cones and was defined as cones that had been burnt heavily enough to cause structural damage to the scales on the cone. The cones were then broken up and the seeds collected, with each of the two burn classes handled separately to avoid mixing of the seeds. The seeds were then stained using tetrazolium to test for any sign of respiration. As all living things respire, respiration was used as a sign of seed viability. Cones were also collected from un-burnt trees located across the road from the burnt area at Flock Hill Station and the seeds were removed to compare the viability rates between seeds from burnt and un-burnt cones.

Results

Ordination

The ordination results show that for each of the vegetation types there was very little change in the species composition. There was very little movement for any of the vegetation types in ordination space and each of the vegetation types held its own space on the ordination with very little overlapping suggesting differences in vegetation composition between the vegetation types (see figure 12 & 13). The area of the ordination where overlapping is occurring is around the centre of the ordination. This is likely due to the exotic pasture grasses recovering in all of the vegetation types (see Table 1 for a table of the five most common species in each vegetation type).

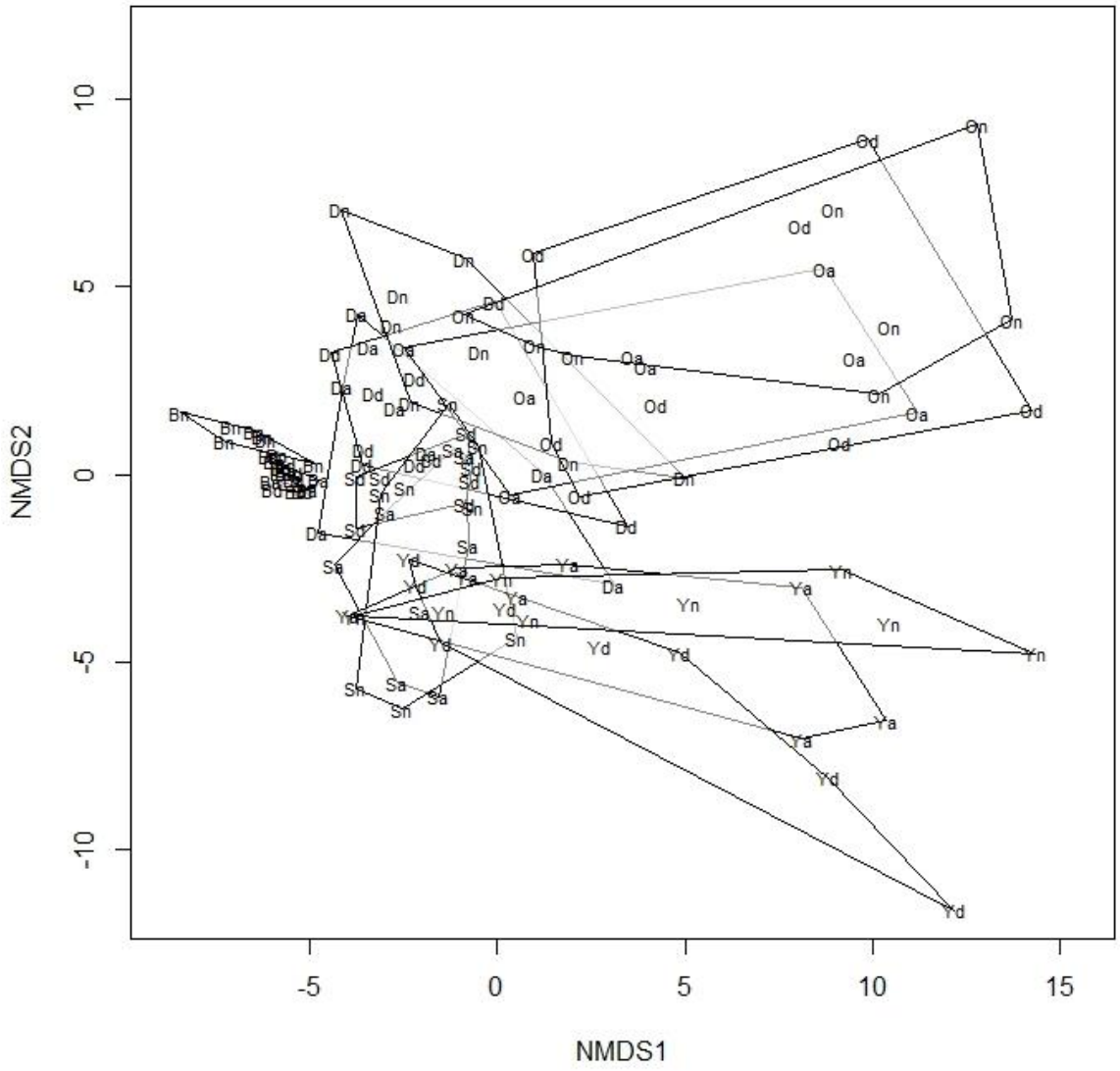

Figure 12. Ordination results for the different vegetation types

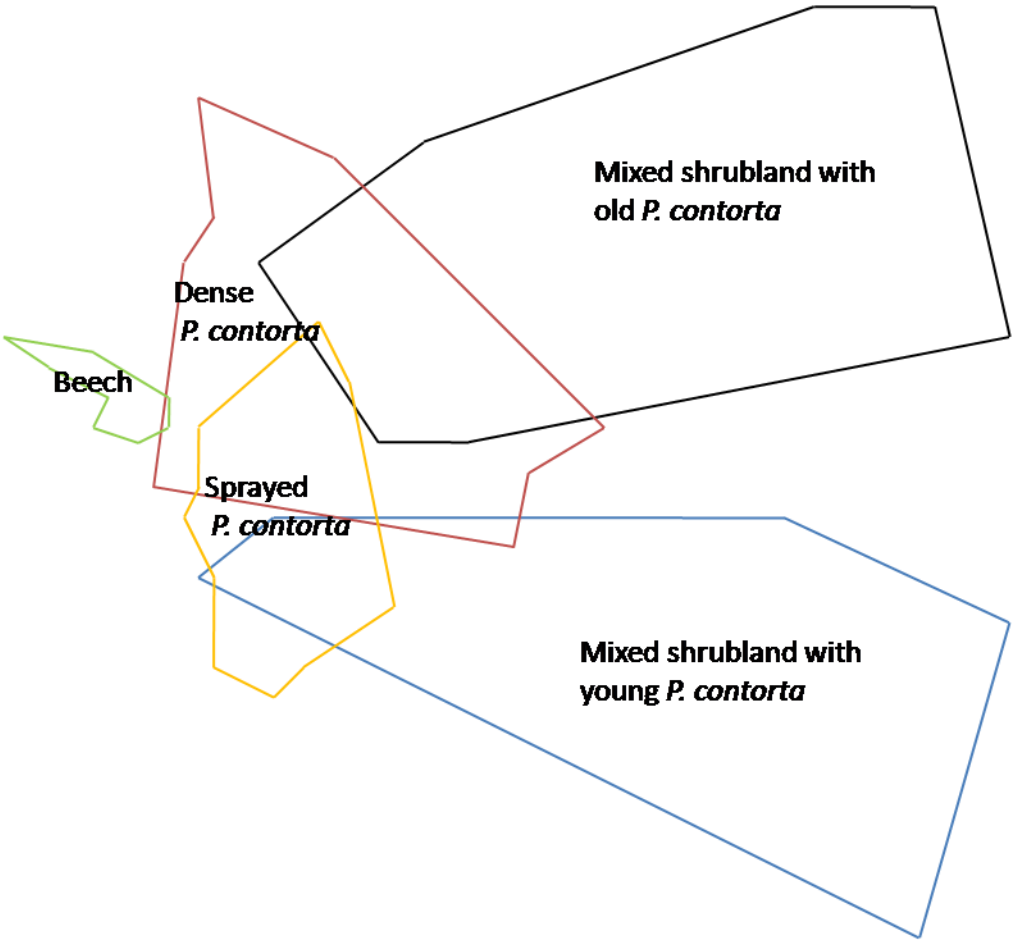

Figure 13. Outline of each vegetation type in the ordination

Table1. Table of the five most common species re-establishing in each vegetation type

Mixed shrubland young contorta	Mixed shrubland old contorta	Sprayed contorta	Dense contorta	beech
Agrostis capillaris	*Agrostis capillaris*	*Agrostis capillaris*	*Agrostis capillaris*	*Agrostis capillaris*
Anthoxanthum odoratum	*Anthoxanthum odoratum*	*Anthoxanthum odoratum*	*Anthoxanthum odoratum*	*Anthoxanthum odoratum*
Pilosella officinarum	*Pilosella Praealtum*	*Festuca rubra*	*Dactylis glomerata*	*Dactylis glomerata*
Pilosella praealtum	*Pilosella officinarum*	*Rumex acetosella*	*Lolium perenne*	*Lolium perenne*
Rumex acetosella	*Trifolium repens*	*Trifolium repens*	*Trifolium repens*	Moss

Vegetation cover

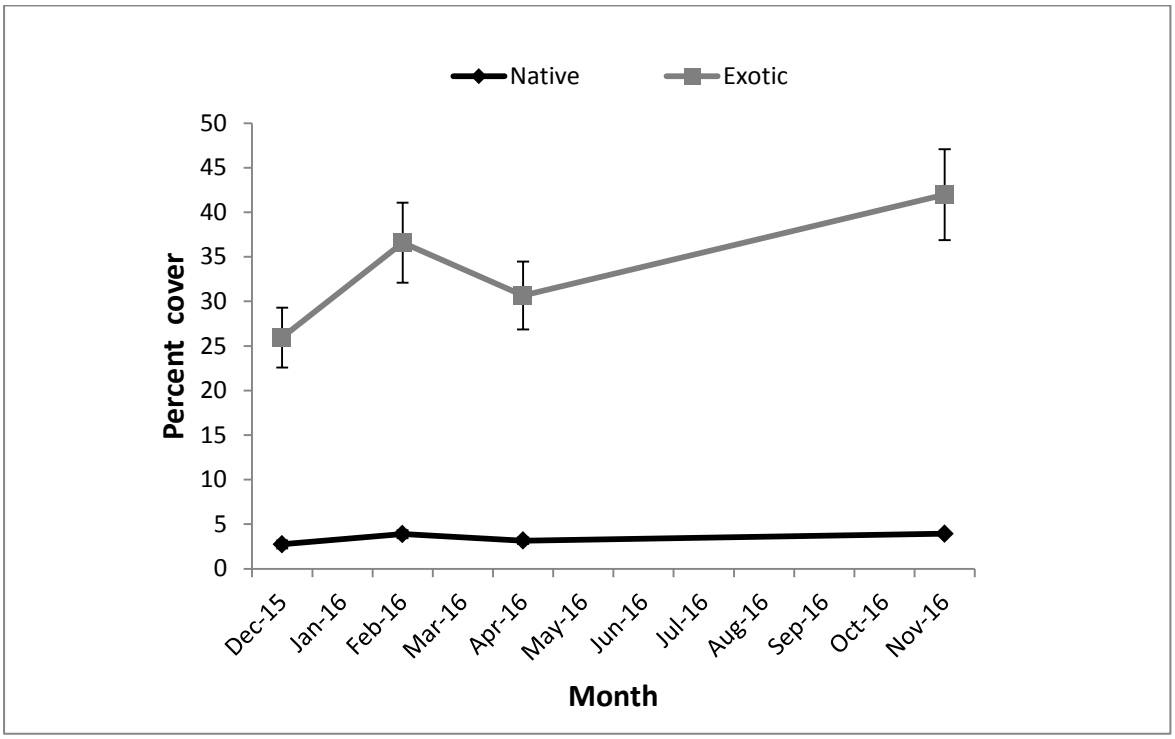

Figure 14. Native and exotic vegetation percentage cover change between December 2015 and November 2016.

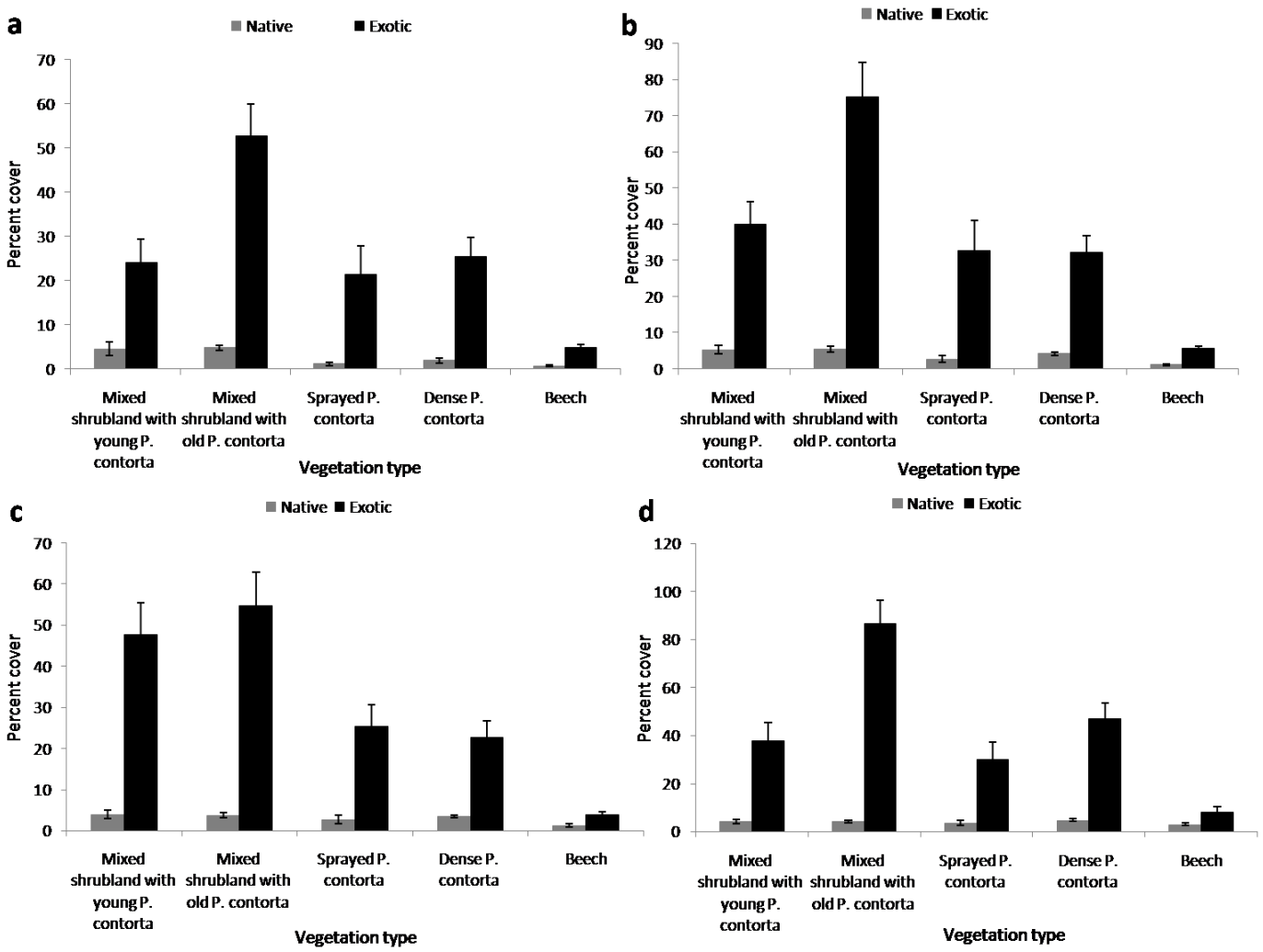

Figure 15. Native and exotic percentage vegetation cover change over time in the different vegetation types. a= December 2015, b= February 2016, c= April 2016, d= November 2016.

Native vegetation

The results show that time since fire had a significant effect on the re-establishment of native species. The results show that there was a significant change in the amount of native cover in the transects over time (P=0.019) (see figure 14). The average native percent cover for native species stayed fairly constant over time (Dec = 2.7±0.45%, Feb = 3.9±0.44%, Apr= 3.2±0.36%, Nov= 3.9±0.33%).The pre-fire vegetation type also had a significant effect on the post-fire recovery of native vegetation (P= < 0.001) (see figure 15). The strongest recovery of native species post-fire occurred in the Dense *P. contorta* vegetation type (all averages given for the different vegetation types are from November 2016) (average cover = 4.8±0.47%), the Native shrubland with young *P. contorta* vegetation type (average cover = 4.1±0.85%) and in the Native shrubland with old *P. contorta* vegetation type (average cover = 4.1±0.38%). The Sprayed *P. contorta* and Beech vegetation types had less recovery of native species with Sprayed contorta having an average native vegetation cover of 3.6±1.2% and Beech having an average native vegetation cover of 3.0±0.5%. Whilst both time and pre-fire vegetation types on their own had a significant impact on the post-fire recovery of

native vegetation, the two combined had no significant difference on the post-fire recovery of the native vegetation at Flock Hill Station (P= 0.335).

Exotic vegetation

Time also had a significant effect on the re-establishment of exotic species. The results show that there was a significant change in the amount of exotic cover in the transects over time (P=0.003) (see figure 14).The average percentage vegetation cover for exotic species increased over time (Dec = 25.9±3.37%, Feb = 36.6±4.49%, Apr= 30.7±3.8%, Nov= 42±5.1%).The pre-fire vegetation type also had a significant effect on the post-fire recovery of exotic vegetation (P= < 0.001) (see figure 15). The strongest recovery of exotic species post-fire occurred in the Native shrubland with old *P. contorta* vegetation type (all averages given for the different vegetation types are from November 2016) (average cover = 86.9±9.4%), the Dense *P. contorta* vegetation type (average cover = 46.9±6.44%) and in the Native shrubland with young *P. contorta* vegetation type (average cover = 37.8±7.63%). The Sprayed *P. contorta* and Beech vegetation types had less recovery of exotic species with Sprayed *P. contorta* having an average exotic vegetation cover of 30±7.29% and Beech having an average exotic vegetation cover of 8.3±1.93%. Whilst both time and pre-fire vegetation types on their own had a significant impact on the post-fire recovery of exotic vegetation, the two combined had no significant difference on the post-fire recovery of the exotic vegetation at Flock Hill Station (P= 0.076).

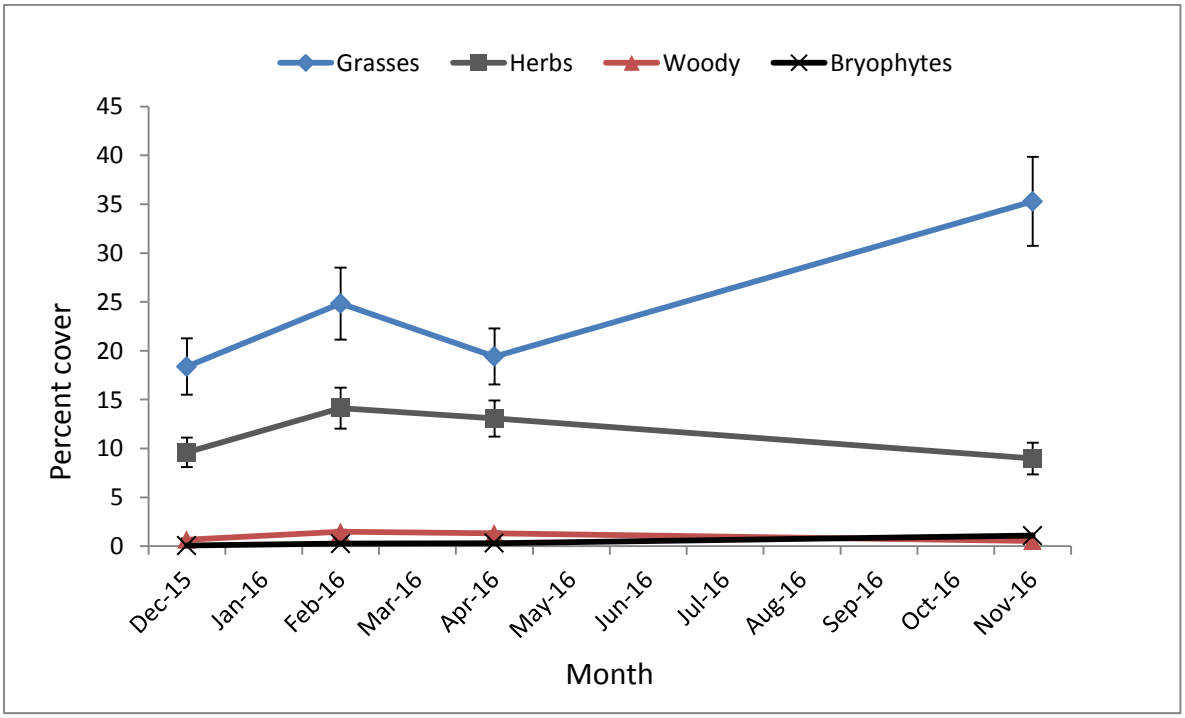

Figure 16. Change in vegetation growth form percent cover between December 2015 and November 2016.

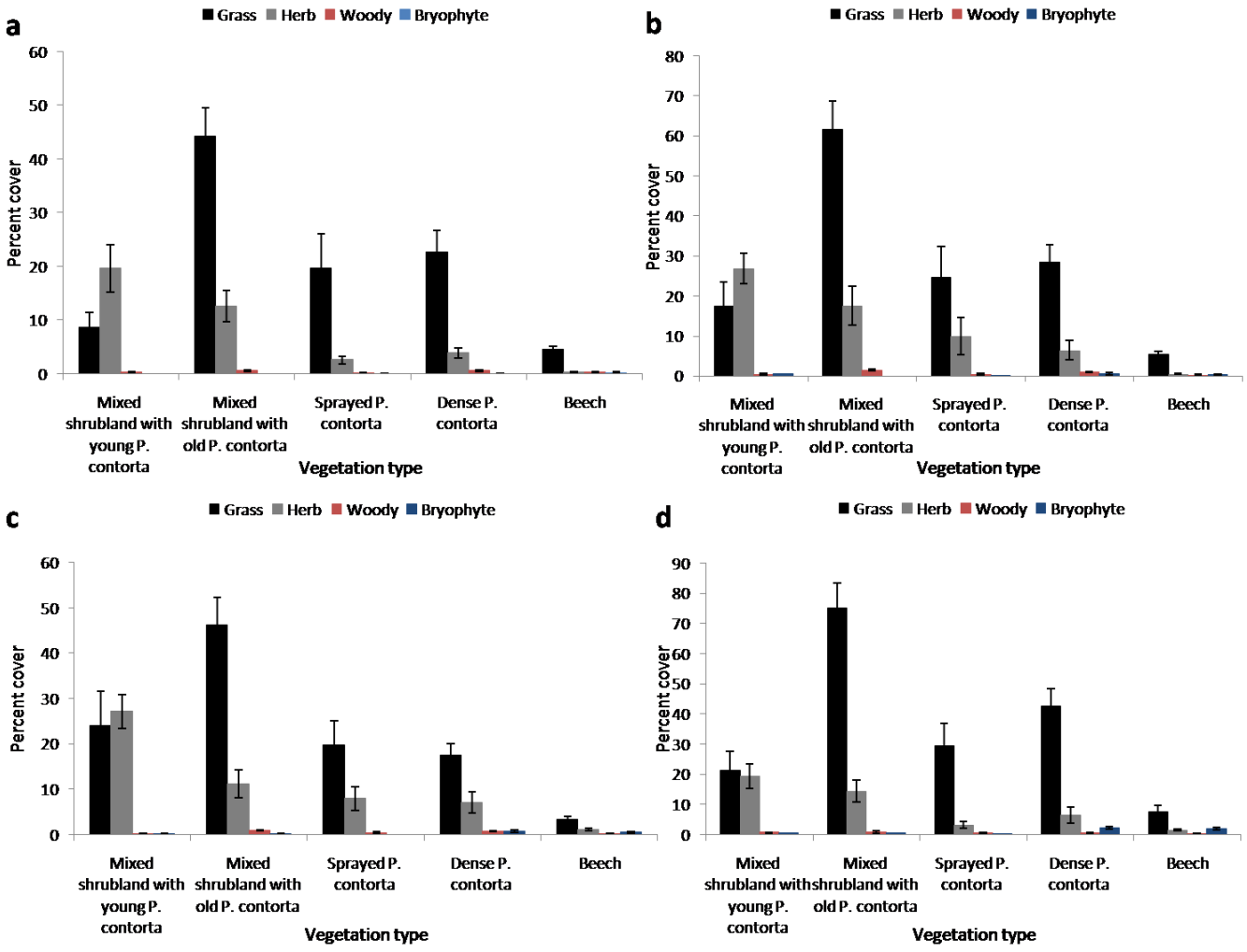

Figure 17. Change in vegetation growth form over time in different vegetation types. a= December 2015, b= February 2016, c= April 2016, d= November 2016.

Grasses

Time since fire had a significant effect on the re-establishment of all grass species (native and exotic). The results show that there was a significant change in the amount of grass cover in the transects over time (P= < 0.001) (see figure 16).The average percentage vegetation cover for all grass species increased over time (Dec = 18.4±2.88%, Feb = 24.8±3.68%, Apr= 19.4±2.86%, Nov= 35.3±4.54%).The pre-fire vegetation type also had a significant effect on the post-fire recovery of all grassy vegetation (P= < 0.001) (see figure 17).The strongest recovery of grass species post-fire occurred in the Native shrubland with old *P. contorta* vegetation type (all averages given for the different vegetation types are from November 2016) (average cover = 75.1±8.24%), the Dense *P. contorta* vegetation type (average cover = 42.6±5.79%) and in the Native shrubland with young *P. contorta* vegetation type (average cover = 29.6±6.15%). The Sprayed *P. contorta* and Beech vegetation types had less recovery of grass species with Sprayed *P. contorta* having an average grassy vegetation cover of 21.4±7.4% and Beech having an average grassy vegetation cover of 7.7±2.01%.Whilst both time and pre-fire vegetation types on their own had a significant

impact on the post-fire recovery of grassy vegetation, the two combined had no significant difference on the post-fire recovery of the grassy vegetation at Flock Hill Station (P= 0.165).

Herbaceous vegetation

Time had no significant effect on the re-establishment of all herbaceous species (native and exotic). The results show that there was no significant change in the amount of herbaceous cover in the transects over time (P=0.080) (see figure 16).The average percentage vegetation cover for herbaceous species initially began to increase but by November 2016 had a lower percentage cover than in December 2015 (Dec = 9.6±1.52%, Feb = 14.1±2.1%, Apr= 13.1±1.85%, Nov= 9±1.61%).The pre-fire vegetation type however, did have a significant effect on the post-fire recovery of herbaceous vegetation (P= < 0.001) (see figure 17).The strongest recovery of herbaceous species post-fire occurred in the Native shrubland with young *P. contorta* vegetation type (all averages given for the different vegetation types are from November 2016) (average cover = 19.4±4.17%), and in the Native shrubland with old *P. contorta* vegetation type (average cover = 14.4±3.62%). The Dense *P. contorta*, Sprayed *P. contorta* and Beech vegetation types had less recovery of herbaceous vegetation with Dense *P. contorta* having an average herbaceous cover of 6.4±2.65%, Sprayed *P. contorta* having an average herbaceous vegetation cover of 3.2±1.1% and Beech having an average herbaceous vegetation cover of 1.5±0.23%.Both time and pre-fire vegetation type combined had no significant impact on the post-fire recovery of the herbaceous vegetation (P= 0.793).

Woody vegetation

Time had a significant effect on the re-establishment of all woody species (both native and exotic). The results show that there was a significant change in the amount of woody vegetation cover in the transects over time (P=0.011) (see figure 16).The average percentage vegetation cover for exotic species increased over time from Dec 2015 to Feb 2016 but by Nov 2016 the woody vegetation cover had dropped to below the Dec 2015 cover (Dec = 0.7±0.13%, Feb = 1.5%±0.2, Apr= 1.3±0.21%, Nov= 0.6±0.1%).The pre-fire vegetation type also had a significant effect on the post-fire recovery of native vegetation (P= < 0.001) (see figure 17).There was low recovery of woody vegetation across all vegetation types (all averages given for the different vegetation types are from November 2016). The Native shrubland with old *P. contorta* vegetation type had an average cover of 0.9±0.32%, the Native shrubland with young *P. contort*a vegetation type had an average cover of 0.6±0.18%, the Dense *P. contorta* vegetation type had an average cover of 0.6±0.15%. The Sprayed *P. contorta* vegetation type had an average cover of 0.5±0.21% and the Beech vegetation type had an average vegetation cover of 0.2±0.09%.Whilst both time and pre-fire vegetation types on their own had a significant impact on the post-fire recovery of woody vegetation, the two combined had no significant difference on the post-fire recovery of the woody vegetation at Flock Hill Station (P= 0.259).

Bryophytes

Time also had a significant effect on the re-establishment of bryophytes. The results show that there was a significant change in the amount of bryophyte cover in the transects over time (P= < 0.001) (see figure 16).The average percentage vegetation cover for bryophytes steadily increased over time (Dec = 0.1±0.04%, Feb = 0.3±0.08%, Apr= 0.4±0.08%, Nov= 1.1±0.18%).The pre-fire vegetation type also had a significant effect on the post-fire recovery of bryophytes (P= < 0.001) (see figure 17). The strongest recovery of bryophitic species post-fire occurred in the Dense *P. contorta* vegetation type (all averages given for the different vegetation types are from November 2016) (average cover = 2.2±0.5%), and in the Beech vegetation type (average cover = 1.9±0.43%). The Native shrubland with young *P. contorta*, Native shrubland with old *P. contorta* and Sprayed *P. contorta* vegetation types had less recovery of bryophytes with Native shrubland with young *P. contorta* having an average exotic bryophyte cover of 0.5±0%, Native shrubland with old *P. contorta* having an average bryophyte cover of 0.5±0%, and Sprayed *P. contorta* having an average bryophyte cover of 0.375±0.08%. Both time and pre-fire vegetation types on their own had a significant impact on the post-fire recovery of bryophytes, the two combined also had a significant difference on the post-fire recovery of the bryophytes at Flock Hill Station (P= 0.0001).

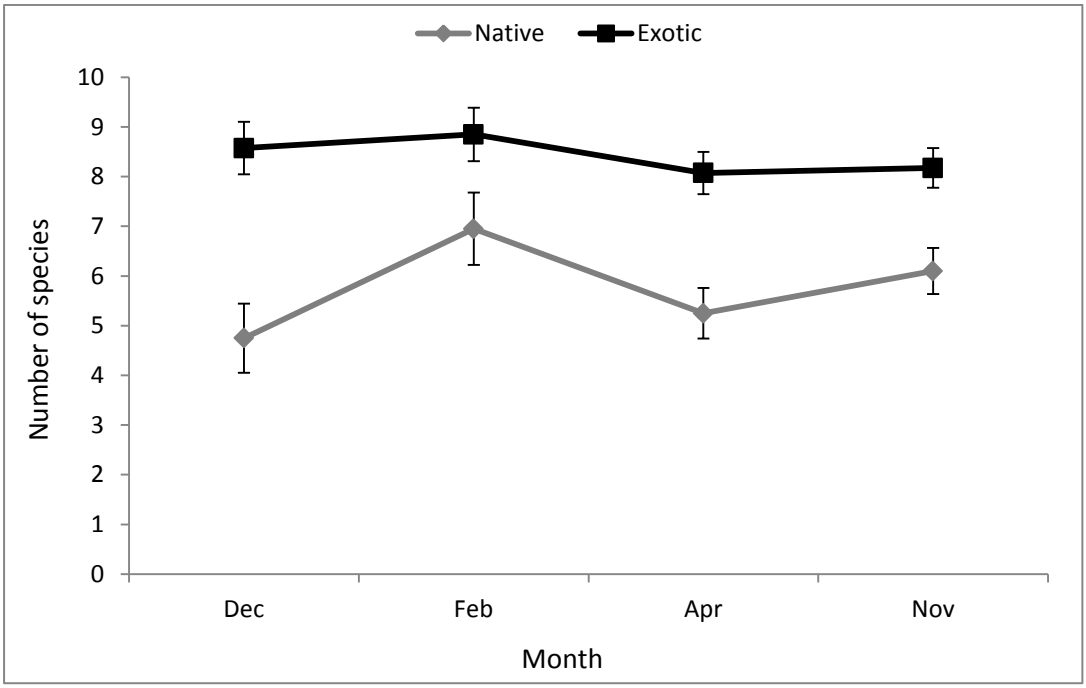

Figure 18. Changes in the number of native and exotic species from December 2015 to November 2016.

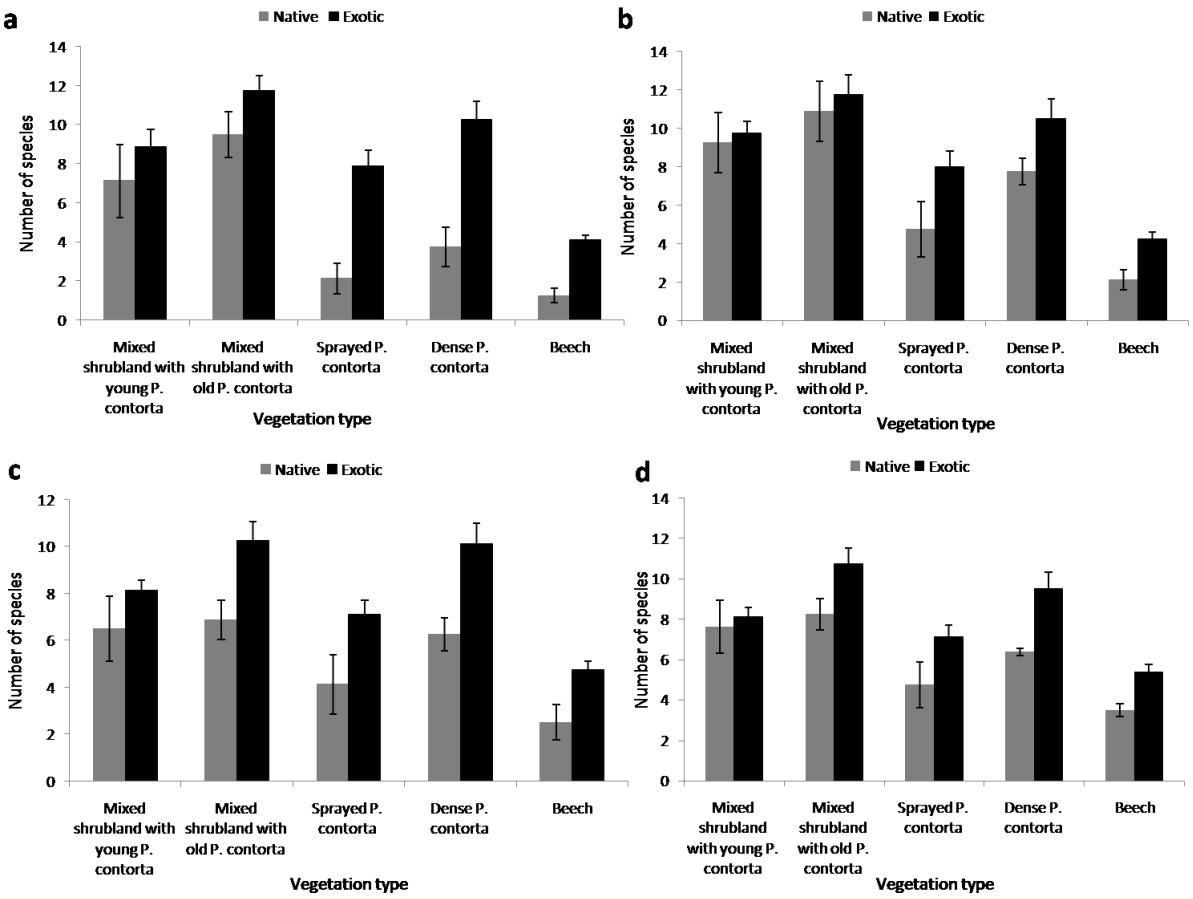

Figure 19. Change in the number of native and exotic species over time in different vegetation types. a= December 2015, b= February 2016, c= April 2016, d= November 2016.

Species Richness

Native vegetation

The results show that time since fire had a significant effect on the number of native species in a transect. The results show that there was a significant change in the number of native species in the transects over time (P=0.009) (see figure 18).The average number of native species per transect across the transects in December 2015 was 4.8±0.7 species per transect. This rose to 7±0.73 species per transect in February 2016 before dropping to 5.3±0.51 species per transect in April 2016. By the start of the second growing season, in November 2016, the number of native species per transect had risen to 6.1±0.46 species per transect. The pre-fire vegetation type also had a significant effect on the number of native species post-fire (P= <0.001) (see figure 19). The average number of native species per transect (all averages given for the different vegetation types are from November 2016) for the Mixed shrubland with young *P. contorta* vegetation type was 7.6±1.32 species per transect, Mixed shrubland with old *P. contorta* vegetation type had 8.3±0.77 native species per transect, Sprayed *P. contorta* vegetation type had 4.8±1.14 native species per transect. The average number of native species per transect for the Dense *P. contorta* vegetation type was 6.4±0.18 species per transect, whilst the Beech vegetation type had 3.5±0.33native

species per transect. Whilst both time and pre-fire vegetation types on their own had a significant impact on the number of native species in a transect, the two combined had no significant effect on the number of the native species in a transect (P= 0.421).

Exotic vegetation

Time had no significant effect on the number of exotic species in a transect. The results show that there was no significant change in the number of exotic species in the transects over time (P=0.292) (see figure 18). The average number of exotic species in a transect for December 2015 was 8.6±0.53 species per transect. For February 2016 the average number of exotic species was 8.9±0.54 species per transect, this dropped slightly to 8.1±0.42 species per transect for April 2016. By the start of the second growing season, in November 2016, the number of exotic species per transect was 8.2±0.4 species per transect. The pre-fire vegetation type also had a significant effect on the number of exotic species post-fire (P= <0.001) (see figure 19).The average number of exotic species per transect (all averages given for the different vegetation types are from November 2016) for the Mixed shrubland with young *P. contorta* vegetation type was 8.2±0.44 species per transect, Mixed shrubland with old *P. contorta* vegetation type had 10.8±0.75 exotic species per transect, Sprayed *P. contorta* vegetation type had 7.1±0.58 exotic species per transect. The average number of exotic species per transect for the Dense *P. contort*a vegetation type was 9.5±0.84 species per transect, whilst the Beech vegetation type had 5.4±0.37 exotic species per transect. The combined influence of time and pre-fire vegetation type had no significant influence on the number of exotic species per transect (p= 0.838).

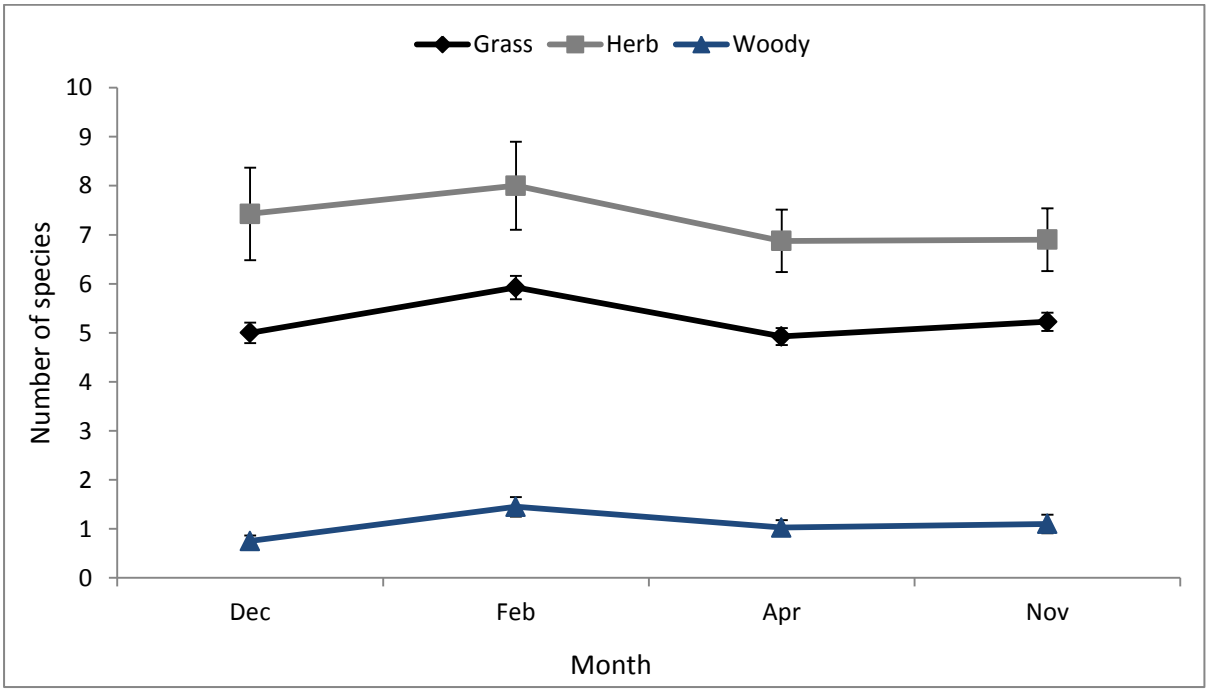

Figure 20. changes in the number of species from December 2015 to November 2016 for different vegetation growth forms.

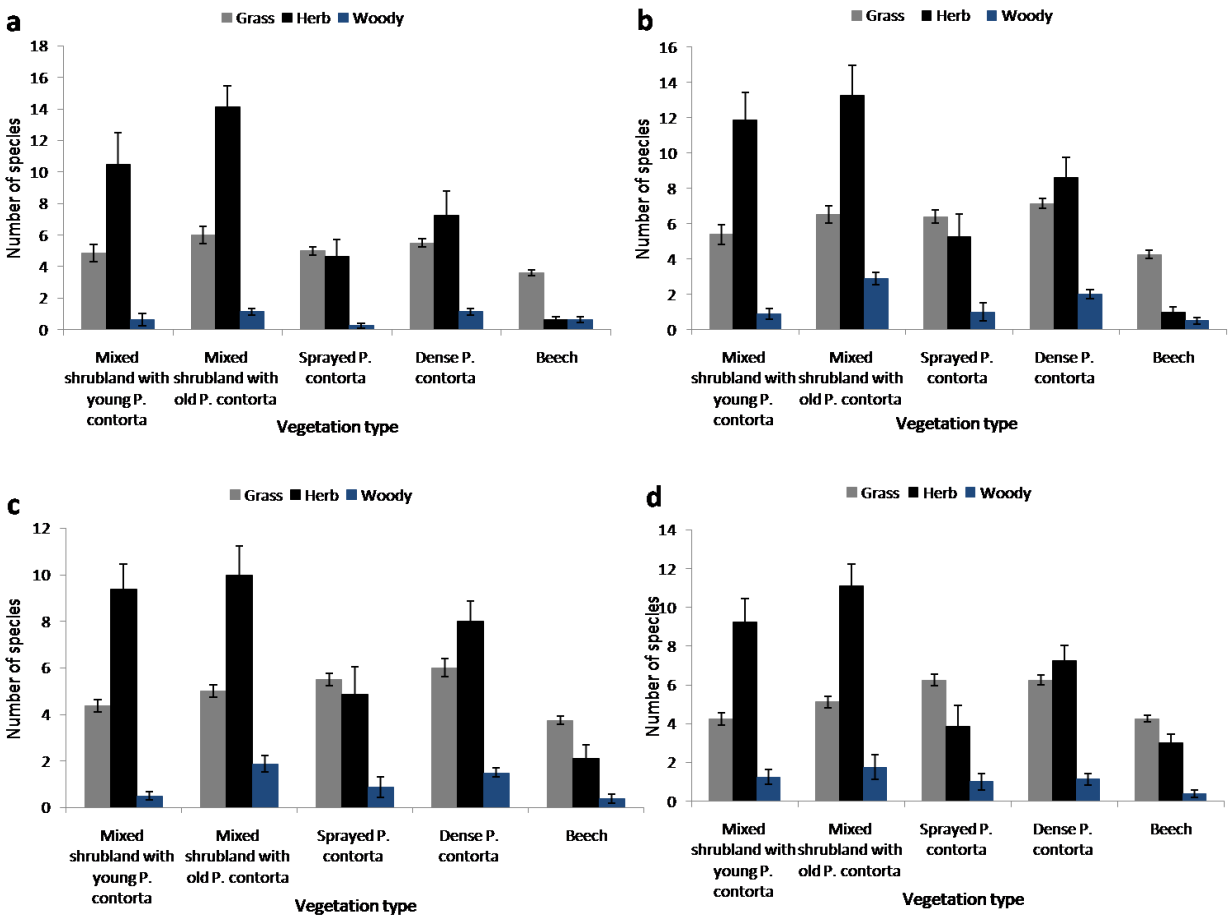

Figure 21. Changes in the number of species for different growth forms. a= December 2015, b= February 2016, c= April 2016, d= November 2016.

Grasses

Time had a significant effect on the number of grass species in a transect. The results show that there was a significant change in the number of grass species in the transects over time (P=<0.001) (see figure 20). The average number of grass species per transect in December 2015 was 5±0.21 species per transect. This rose to 5.9±0.24 species per transect in February 2016 before dropping to 4.9±0.17 species per transect in April 2016. By the start of the second growing season, in November 2016, the number of grass species per transect had risen to 5.2±0.18 species per transect. The pre-fire vegetation type also had a significant effect on the number of grass species post-fire (P= <0.001) (see figure 21).The average number of grass species per transect (all averages given for the different vegetation types are from November 2016) for the Mixed shrubland with young *P. contorta* vegetation type was 4.3±0.31 species per transect, Mixed shrubland with old *P. contorta* vegetation type had 5.1±0.29 grass species per transect, Sprayed *P. contorta* vegetation type had 6.3±0.31 grass species per transect. The average number of grass species per transect for the Dense *P. contorta* vegetation type was 6.3±0.25 species per transect, whilst the Beech vegetation type had 4.3±0.16 grass species per transect. Whilst both time and pre-fire vegetation types on their own had a significant impact on the number of grass species in a transect, the two

combined had no significant difference on the number of grass species in a transect (P= 0.117).

Herbaceous vegetation

Time had no significant effect on the number of Herbaceous species in a transect. The results show that there was no significant change in the number of Herbaceous species in the transects over time (P=0.377) (see figure 20). The average number of herbaceous species in a transect for December 2015 was 7.4±0.94 species per transect. For February 2016 the average number of herbaceous species was 8±0.9 species per transect, this dropped to 6.9±0.64 species per transect for April 2016. By the start of the second growing season, in November 2016, the number of herbaceous species per transect was 6.9±0.64 species per transect. The pre-fire vegetation type also had a significant effect on the number of herbaceous species post-fire (P= <0.001) (see figure 21). The average number of herbaceous species per transect (all averages given for the different vegetation types are from November 2016) for the Mixed shrubland with young *P. contorta* vegetation type was 9.3±1.2 species per transect, Mixed shrubland with old *P. contorta* vegetation type had 11.1±1.1 herbaceous species per transect, Sprayed *P. contorta* vegetation type had 3.9±1.04 herbaceous species per transect. The average number of herbaceous species per transect for the Dense *P. contorta* vegetation type was 7.3±0.77 species per transect, whilst the Beech vegetation type had 3±0.46 herbaceous species per transect. The combined influence of time and pre-fire vegetation type had no significant influence on the number of herbaceous species per transect (p= 0.420).

Woody vegetation

Time had a significant effect on the number of woody species in a transect. The results show that there was a significant change in the number of woody species in the transects over time (P=<0.011) (see figure 20).The average number of woody species in a transect for December 2015 was 0.8±0.12 species per transect. This rose to 1.5±0.2 species per transect in February 2016 before dropping to 1±0.15 species per transect in April 2016. By the start of the second growing season, in November 2016, the number of woody species per transect was 1.1±0.19 species per transect. The pre-fire vegetation type also had a significant effect on the number of woody species post-fire (P= <0.001) (see figure 21). The average number of woody species per transect (all averages given for the different vegetation types are from November 2016) for the Mixed shrubland with young *P. contorta* vegetation type was 1.2±0.36 species per transect, Mixed shrubland with old *P. contorta* vegetation type had 1.8±0.65 woody species per transect, Sprayed *P. contorta* vegetation type had 1±0.42 woody species per transect. The average number of woody species per transect for the Dense *P. contorta* vegetation type was 1.1±0.29 species per transect, whilst the Beech vegetation type had 0.4±0.18 woody species per transect. Whilst both time and pre-fire vegetation types on their own had a significant impact on the number of woody

species in a transect, the two combined had no significant difference on the number of woody species in a transect (P= 0.259).

Re-sprouting in *Discaria toumatou*

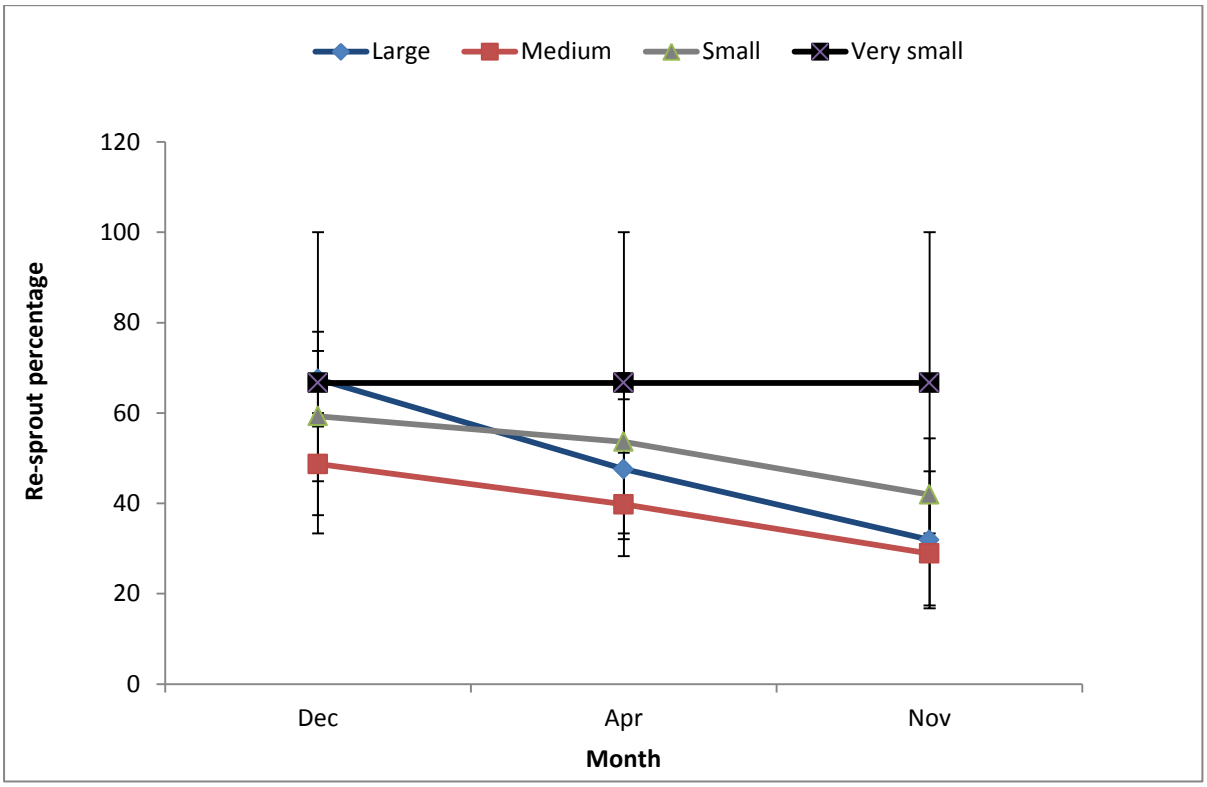

Figure 22. Re-sprouting percentage for *Discaria toumatou* from December 2015 to November 2016.

Time had no significant effect on the survival of matagouri re-sprouting following fire (P= 0.130) (see figure 22). The results also show that plant size has no significant influence on matagouri re-sprouting and survival post fire (P= 0.296) (see figure 22). Time had no impact on the survival of the very small matagouri size class re-sprouts with no fatalities from December 2015 through to November 2016. The other three size classes however did suffer re-sprout fatalities from December 2015 through to November 2016. The very small size class had a re-sprouting percentage of 66.6±33.3% in December 2015 and was still 66.6±33.3% in November 2016. The small size class had a re-sprouting percentage of 59.3±14.4% in December 2015 but had dropped down to 41.9±12.5% in November 2016. The medium size class had a re-sprout percentage of 48.7±11.3% in December 2015 but had dropped to 28.9±11.4% in November 2016. The large size class had the highest re-sprouting percentage in December 2015 of 67.5±10.5% but by November 2016 it had dropped down to 31.9±15.2%. both time and plant size individually had no significant impact on re-sprouting in matagouri and this is the same for the two combined, with time and plant size combined having no significant influence on matagouri re-sprouting (P= 0.984). Other woody plants observed to be able to re-sprout following fire include *Aristotelia fruticosa* and *Coprosma propinqua* (see figures 23, 24 & 25), however as there were so few individuals

present and these were only located in two transects it was not possible to analyse them (see table 2 for list of all woody species present in re-sprout transects).

Table 2. List of all woody species present in the transects

Species	Total number present
Leptospermum scoparium	1050
Pinus contorta	1541
Dracophyllum acerosum	50
Ozothamnus leptophyllus	28
Hebe subalpina	9
Fuscospora cliffortioides	506
Discaria toumatou	262
Coprosma propinqua	11
Aristotelia fruticosa	3

Figure 23. Re-sprouting *Discaria toumatou*

Figure 24. Re-sprouting *Aristotelia fruticosa*

Figure 25. Re-sprouting Coprosma propinqua

Seed viability in *Pinus contorta*

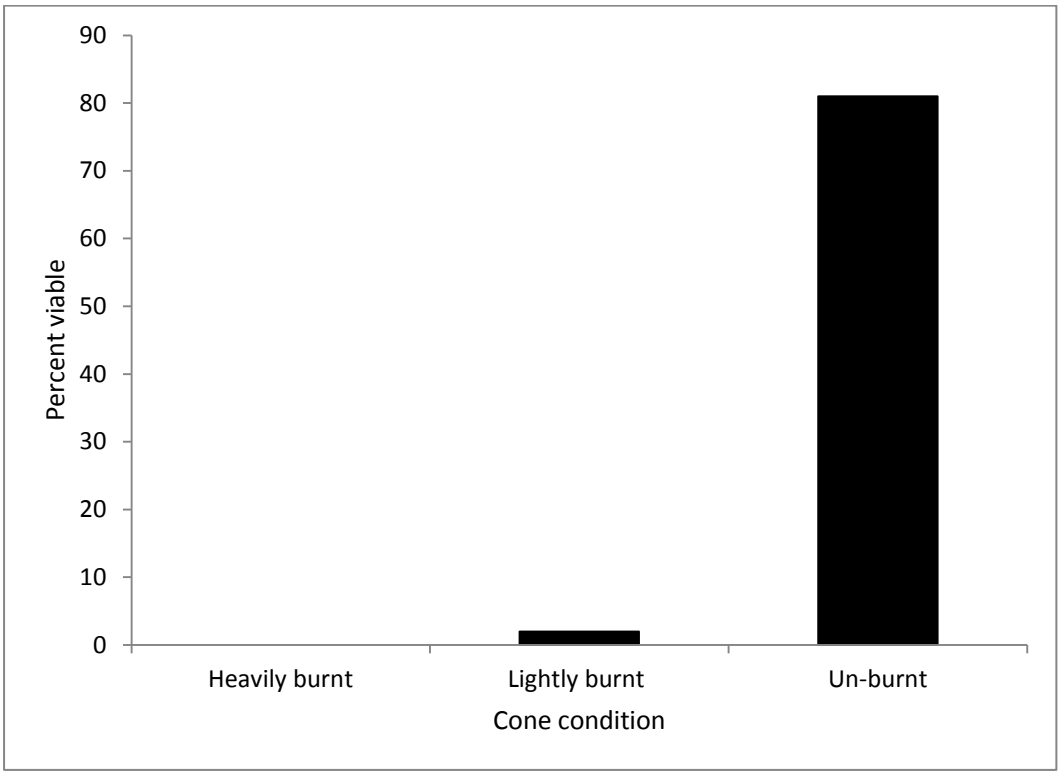

Figure 26. Viability of *P. contorta* seeds from cones with different burn severities.

The viability of *P. contorta* seeds is greatly influenced by fire. Heavily burnt cones had 0% viable seeds, lightly burnt cones had 2% viable seeds and un-burnt cones had 81% viable seeds (see figure 26). The average diameter of the heavily burnt cones was 11.3cm with an average length of 4.69cm. The average number of seeds per cone for the heavily burnt cones was 9.76 seeds per cone. The average diameter of the lightly burnt cones was 12.06cm with an average length of 5.05cm. The average number of seeds per cone for the lightly burnt cones was 2.51 seeds per cone.

Mānuka seed sowing

There was mānuka seed germination in five of the forty plots. Two of the Dense *P. contorta* plots had five mānuka seedlings each and three of the Beech plots had mānuka germinate as well with three, five and eight seedlings. Unfortunately none of the seedlings survived through to the second growing season.

Photo monitoring

Photo monitoring provided a useful visual aid to show how the vegetation recovered in each of the five vegetation types. The photos show the strong vegetation recovery from December 2015 to November 2016 for the Mixed shrubland with young *P. contorta*, Mixed shrubland with old *P. contorta*, Sprayed *P. contorta*, and dense *P. contorta* vegetation types (see figures 27 - 34). The Beech vegetation type showed very little vegetation recovery from December 2015 through to November 2016 (see figures 35 & 36).

Figure 27. A transect from the Mixed shrubland with young *P. contorta* vegetation type from December 2015.

Figure 28. The same transect as figure 27 in November 2016

Figure 29. A transect from the Mixed shrubland with old *P. contorta* vegetation type from December 2015.

Figure 30. The same transect as figure 29 in November 2016

Figure 31. A transect from the Sprayed *P. contorta* vegetation type from December 2015.

Figure 32. The same transect as figure 31 in November 2016

Figure 33. A transect from the Dense *P. contorta* vegetation type from December 2015.

Figure 34. The same transect as figure 33 in November 2016

Figure 35. A transect from the Beech vegetation type from December 2015.

Figure 36. The same transect as figure 35 in November 2016

Discussion

Pinus contorta recovery

This study predicted that *P. contorta* would be the dominant species re-establishing after the fire, however this was not the case and very little recovery of *P. contorta* was observed in the transects. There are likely to be multiple factors affecting the recovery of *P. contorta*, including the fire itself, drought, grazing and the amount of seed entering the burn site. One possible explanation for the lack of recovery in *P. contorta* following the 2015 fire at Flock Hill Station may have been the very dry weather conditions. The weather conditions following the January 2015 fire were characterised by below average rainfall from July 2015 through to April 2016 with the exception of one month (see figure 37). This meant that for almost the entire first growing season the site was experiencing drought conditions. This explanation is supported by other studies. Petrie *et al*. (2016) found that *P. contorta* seedlings are more susceptible to dry conditions than *P. ponderosa* another very common North American pine species. Engelmark *et al*. (2001) list drought as one main reason for seedling loss of *P. contorta* in Swedish forests and believe that *P. contorta* is more likely to invade wetter sites than the native scots pine (*P. sylvestris*).Coops & Waring (2011) investigated how future climate change might affect *P. contorta* distribution through modelling projected climatic changes in the Pacific Northwest. They found that future climate change and especially droughts are likely to be unfavourable for *P. contorta* and favour more drought-adapted species in the area including ponderosa pine, Douglas-fir and western larch. They believe that by the end of the 21st century *P. contorta* may become absent from large areas of its current range due to increased adult mortality. However not all studies support the idea that drought impacts heavily on *P. contorta* regeneration. Harvey *et al.* (2016) looked into how predicted increased wildfires and drought with future climate change may affect North American forests by analysing the post-fire establishment of seedlings in plots across a range of climatic conditions. Harvey *et al.* (2016) believe that future post-fire drought conditions are likely to favour *P. contorta* over other subalpine tree species due to its higher drought tolerance. One difference between the studies of Coops & Waring (2011) and Harvey *et al.* (2016) is that Coops & Waring (2011) did not include fire in

their modelling of future climate change, whereas the Harvey *et al.* (2016) was looking at the effects of drought post-fire.

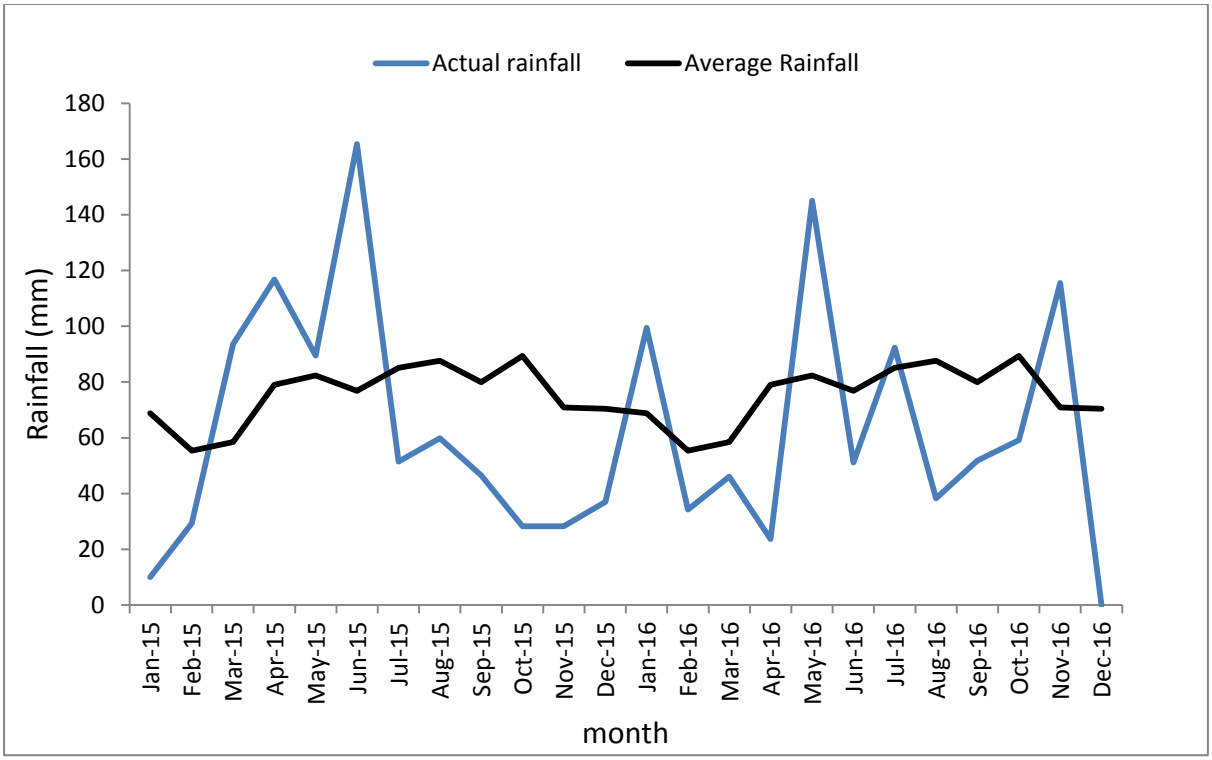

Figure 37. The average rainfall vs. actual rainfall from January 2015 to November 2016 (NIWA data).

The fire itself is likely to be a reason for the lack of *P. contorta* regeneration. This study found that *P. contorta* suffered very high seed mortality in the fire (see figure 26). This would have greatly reduced the number of viable seeds entering the burn site from the burnt trees during and after the fire. These results however are not supported by the majority of the literature. The literature supports the idea that *P. contorta* is a serotinous species and that *P. contorta* should re-establish following fire (Muir & Lotan 1985, Turner *et al*. 2007, Pierce & Taylor 2011, Strong & Hills 2013, Edwards *et al*. 2015). However these studies were carried out in *P. contorta*'*s* native range of North America. The *P. contorta* present in New Zealand came from small subsets of the North American population (Miller & Ecroyd 1987). The *P. contorta* at Craigieburn and Flock Hill is the coastal variety *Pinus contorta* var. *contorta* which is far more vigorous than the inland varieties of *P. contorta* but is also far less serotinous than the inland varieties of *P. contorta* (Miller & Ecroyd 1987, N. Ledgard pers. comm. February 2017). Ledgard (2001) also suggests that regular burning

carried out by farmers help to control wilding *P. contorta* in the New Zealand high country and that the spread of *P. contorta* since the 1950s is partly due to restrictions to burning. This could help to explain the why there was such a low viability observed in the seed viability test carried out on the *P. contorta* seeds extracted from the burnt cones and also the lack of *P. contorta* regeneration following the fire.

It is important to remember that *Pinus contorta* is a pioneer species (Ledgard 2001, Ledgard & Paul 2008, Ledgard 2009) and that the coastal variety of *P. contorta* present at the study site is a vigorous spreader (Miller & Ecroyd 1987) that would be well suited to spreading in the conditions created following the fire. However extensive control work has been carried out at Flock Hill Station and the surrounding areas and it is likely that this extensive wilding control work would have influenced the regeneration of *P. contorta*. In the months following the fire the surviving trees around the perimeter of the burn site were felled. This would have stopped the trees from being able to drop seeds into the burn site after the summer. Felling and spraying operations were also carries out across Flock Hill Station in the late winter and into the spring following the fire to remove a large majority of the remaining mature wilding pines, most of which were *P. contorta*, from the surrounding area. Spraying and felling had also been carried out at Helicopter Hill a few years prior to the fire. Helicopter Hill is a site in the foothills of the Craigieburn Range that was extensively covered in *P. contorta* and would have been a major source of *P. contorta* seeds arriving onto Flock Hill Station due to it being upwind but in close proximity to the study site. The removal of these wilding pines has undoubtedly helped to reduce the number of *P. contorta* seeds entering the study site.

Grazing is likely to also be a major factor for the lack of recovery from *P. contorta* following the fire. Non-domestic grazers, such as rabbits, hares, possums, and deer, were present at the burn site for the entire length of the study, while domestic sheep and cattle were present from early January 2016 through the summer, autumn, and winter but had been removed by November 2016. The idea that grazing is likely to have had a large impact on *P. contorta* regeneration is supported by other studies. Ledgard & Norton (2008) looked into the impacts that grazing animals have on wilding conifers in New Zealand's South Island high country and found that sheep and rabbits have a significant effect on the establishment of wilding conifer species, including *P. contorta.* They suggest that increased rabbit controls

are likely to be a significant reason for the increase in wilding conifers in New Zealand. Ledgard & Norton (2008) found that grazing tends to kill the majority of conifers soon after they had germinated. This would have been the stage that the *P. contorta* would have been at after the fire and could be one likely cause for the low *P. contorta* numbers re-establishing after the fire.

Unlike Ledgard & Norton (2008) who looked at the direct effects of grazing on *P. contorta* seedling establishment, Norman & Taylor (2005) looked at the historical effects of fire and grazing on pine establishment in meadows in North-Eastern California, USA. Norman & Taylor (2005) found that historic fire and grazing have prevented conifers, including *P. contorta*, from establishing in the meadows and that the expansion of conifers into these meadows is likely due to a reduction in sheep numbers. Engelmark *et al*. (2001) list animal grazing as one of the main reasons for the loss of *P. contorta* within the first year of establishment in the Swedish forest. Other studies have also been carried out looking at the effects of grazing on pines. Kingery & Graham (1991) looked at the effects of cattle grazing on *Pinus ponderosa* regeneration and found that cattle reduced the numbers of *P. ponderosa* mainly through trampling rather than through grazing. While Boulant *et al*. (2008) found that grazing reduced the numbers of the invasive *Pinus nigra* in France. Grazing has been well known to reduce the success of establishing woody vegetation for centuries (Humphrey & Patterson 2000, Esteban, De Palacios & Rodriguez-Losada Aguado 2010, Kouba *et al*. 2015). In Europe, fire followed by grazing, has been used in agriculture to control conifers for millennia (Schworer *et al*. 2015). Fire, followed by grazing, has been used by European farmers for thousands of years to remove conifers from the conifer dominated sub-alpine areas of Europe to create alpine meadows throughout alpine Europe (Vittoz *et al*. 2008, Hejcman *et al*. 2013, Schworer *et al*. 2015) and that in areas where fire and grazing has ceased the conifers begin to establish once more (Tasser & Tappeiner 2002). Fire and grazing was also widely used throughout the conifer dominated forests of Scandinavia for millennia to remove the conifers for livestock grazing in the resulting meadows that formed post fire (Eriksson, Cousins & Bruun 2002, Dahlstrom, Cousins & Eriksson 2006, Berglund *et al*. 2014).

It is likely that the combined effects of fire, grazing and the extensive wilding control work that has been undertaken in the surrounding areas has resulted in the low numbers of

P. contorta re-establishing following the fire. The fire likely killed the majority of seeds in the cones, whilst the control work would have greatly reduced the amount of seeds entering the burnt area and grazing animals would have strongly impacted the survival of any germinating seeds. The dry weather conditions also likely played a role as well.

Native vegetation recovery

This study predicted that the native vegetation would not be able to re-establish following the fire as New Zealand's native flora had evolved largely in the absence of fire. This was largely supported by the results presented here. The native vegetation was not able to match the re-establishing exotic vegetation (see figure 14). The exotic vegetation recovery was driven by the strong recovery of exotic pasture grasses such as browntop (*Agrostis capillaris*) and sweet vernal (*Anthoxanthum odoratum*) and herbaceous weeds such as sheep's sorrel (*Rumex acetosella*) and the hawkweed's (*Pilosella officinarum*, *Pilosella praealtum* and *Hieracium lepidulum*) which all re-established very quickly after the fire. The over-sown pasture grasses and herbs (*Trifolium repens*, *Lolium perenne*, and *Dactylis glomerata*) all established well in the shaded sites of the dense *P. contorta* and beech vegetation types. Some native species showed the ability to re-establish or re-sprout after fire, including the herbaceous species of *Viola cunninghamii* and *Wahlenbergia albomarginata*. The only native woody shrub or tree species to show any widespread ability to recover following fire was *Discaria toumatou* which re-sprouted following fire. *Coprosma propinqua* and *Aristotelia fruticosa* also showed some ability to re-sprout but were not able to be assessed fully as they were present in very small numbers. One native woody species showed the ability to regenerate from seed and that was *Ozothamnus leptophyllus,* however this was only in small numbers. There are several possible reasons for the lack of native vegetation recovery; they include the lack of fire adaptability, the dry weather conditions and grazing.

The lack of adaptations to fire from the New Zealand flora could be one reason for the low levels of recovery from the native vegetation following the fire. The idea of New Zealand's flora not being adapted to fire is supported by the literature. Wardle (2001) and Ogden *et al*. (2003) looked through the charcoal records and found that historic Māori fires

in New Zealand led to a reduction in native vegetation in those areas burnt. Ogden *et al*. (2003) believe that this burning would have increased the numbers of bracken (*Pteridium esculentum*). Bracken was observed in some of the transects at the study site following the fire. The bracken likely re-sprouted from underground rhizomes that survived the fire. Guild & Dudfield (2010) found that fires lit by European settlers to convert the land to pasture for grazing was highly effective in killing the native vegetation, especially in the dry Canterbury region. Perry *et al*. (2014) believe that the New Zealand flora is poorly adapted to survive fire and that New Zealand has very few fire adapted species. Perry *et al*. (2014) name Mānuka (*Leptospermum scoparium*) and matagouri (*Discaria toumatou*) as two fire adapted species. Bond *et al*. (2004) also describe mānuka as a fire adapted species however our study found no evidence of mānuka being able to recover following fire either through re-sprouting or through natural regeneration from seed. Bellingham (1998) also describes matagouri as being fire adapted stating that following fire matagouri often produces re-sprouts. The results from this study support this with matagouri of different sizes re-sprouting, with some size classes having as high as 67% re-sprouting in the first growing season following the fire.

The dry weather conditions at the study site likely had an influence on the recovery of the native vegetation. The long period of dry weather experienced at the study site is likely to be one of the reasons for the lack of germination from the mānuka that was sown beside each of the transects. All of the transects where the mānuka did germinate were locations shaded by standing dead trees and it is likely that this shading helped to keep the soil a little more moist. The dry weather conditions are also likely to be one of the reasons for the decline in *D. toumatou* re-sprouts (see figure 37). The dry weather conditions experienced at Flock Hill Station would have effected both the native and exotic vegetation. The dry weather conditions are likely to have played an important role in the reduction in cover for both the native and exotic vegetation from February 2016 to April 2016 (see figures 14 & 16).

Grazing is likely to be another factor affecting the recovery of native vegetation. Non-domestic grazers, such as rabbits, hares, possums, and deer, were present at the burn site for the entire length of the study, while domestic sheep and cattle were present from early January 2016 through the summer, autumn, and winter but had been removed by

November 2016. Evidence of grazing was found on the majority of both the alive and dead re-sprouts of *D. toumatou*. Grazing on *D. toumatou* has also been observed in other studies. Primack (1978) observed that grazing regularly occurred on *D. toumatou* while Bellingham (1998) believes that a reduction in burning and rabbit control has led to an increase in *D. toumatou* since the 1950s. Williams, Kean & Buxton (2010) believe that *D. toumatou* is susceptible to grazing while its shoots are young and have not developed spines and that after *D. toumatou* is burnt the new, spineless shoots are again exposed to grazing. Grazing would not have only effected the native vegetation, grazing would have also had an effect on the exotic vegetation. Grazing would be the likely reason for the reduction, and in some instances, the disappearance of some exotic woody weed species such as scotch broom (*Cytisus scoparius*) and sweet brier (*Rosa rubiginosa*) (Popay & Field 1996, Bellingham & Coomes 2003, Sage, Norton & Espie 2009, Williams, Kean & Buxton 2010). Grazing likely also played a part in the reduction in cover for both native and exotic vegetation from February 2016 to April 2016 (see figures 14 & 16).

It is likely that fire, dry weather conditions, and grazing all played a role in the relatively low recovery of the native vegetation following the fire. I do not believe that competition from exotic species played a large role in the low levels of recovery in the native vegetation as although the exotic pasture grasses covered large areas of the burn site, there was still plenty of areas of bare ground lacking exotics.

Implications of the research

The results of this research suggest that in certain situations fire may be an option for the control of wilding pines. In situations where wilding pines have occupied large areas of an agricultural landscape and removing the pines and returning the site to pasture is the objective then fire may be a useful tool for achieving this. However, for this to be successful it is likely that the site would have to be over-sown and grazed and decent levels of livestock would have to be maintained following the fire to keep the wilding pine from re-establishing on the site. In contrast, for areas where wilding pines are occupying land with some level of conservation value then fire would be a poor tool to use as in this study it was observed that New Zealand's native woody vegetation, with the exception of *Discaria toumatou*, does not respond well to fire and burning would likely do more harm than good. Also there would

likely be far fewer grazers on conservation land and it could be that this would allow wilding *P. contorta* to re-establish. It is also important to remember that fire is very difficult to control and can therefore rapidly become out of control and is therefore always going to be a risky option for controlling wilding pine where other values (native vegetation) are in close proximity. It is always important to assess the risks when considering fire as an option for wilding control and it may likely be that other methods may be more appropriate.

Drawbacks and future research

The two large drawbacks of this study where the over-sowing of pasture grasses and the introduction of livestock to the burnt area. It was not brought to our attention that the study site had been over-sown with grass seed until after the study had been planned. Livestock were not meant to be introduced to the study site for the length of the study, however, in early January 2016 sheep and cattle where present. This resulted in the herbivore exclusion work not being able to be completed as the exclusion cages were not able to handle cattle trampling. The grazing cattle also had a large impact on the vegetation recovery in the beech forest as they trampled the loose ground and pulled some of the establishing grasses out by their roots. This resulted in very little vegetation recovery in the beech forest. Both the over-sowing of grass seed and the introduction of livestock added unexpected dimensions to the study. It should also be mentioned that the results of this study only look at the first growing season and the early stages of the second growing season following the fire. This short duration of the study makes it difficult to make predictions for the medium and long term future of the wilding *Pinus contorta* at Flock Hill Station.

Any future research at Flock Hill Station could look at the recovery of *P. contorta* and the native woody vegetation. As *P. contorta* is a major weed species in the area of the study and as this study only focussed on the first growing season and the early stages of the second growing season, research looking into the medium and long term recovery of *P. contorta* would be useful in determining whether the January 2015 fire has had any long term impacts on the invading *P. contorta*. Continued research into the recovery of the native woody vegetation would also be useful over the medium to long term to see if the re-sprouting *D. toumatou* survive and whether *L. scoparium* can establish on the site. It would

also be interesting to look into whether the burnt areas of beech can recover or whether they will become dominated by exotic species after the fire.

Conclusions

It appears that fire followed by grazing has a large negative impact on *P. contorta* recovery. Fire followed by grazing also appears to have a negative impact on the recovery of the native woody vegetation; however *D. toumatou* has shown that it is capable of recovering following fire through being able to re-sprout. The dry weather conditions experienced at Flock Hill Station over the period of the study would have also had an effect on the recovering vegetation. Fire does not appear to have a large impact on exotic pasture grasses, with exotic pasture grasses showing strong recovery following the fire.

Fire may be a useful management tool for wilding *Pinus contorta* in agricultural landscapes, especially if it is the less serotinous coastal variety of *P. contorta*. In agricultural systems where a large area of land is in *P. contorta* and is planned to be turned back to pasture and grazing can be maintained in moderate levels then fire could be considered as an option based off of the results of this study.

With climate change it is likely that fire will become more frequent in the eastern South Island as it becomes warmer and drier (Perry *et al*. 2014). This is likely to result in more fires in the Canterbury high country. Already in 2017 there have been two fires around the Flock Hill area with one at Cora Lynn Station and a large fire at Broken River. Previous fires in the area have occurred at Cora Lynn and Waterfall Terrace under Mt Horrible, but both were some years ago, supporting the idea that fire frequency is increasing.

Fires in the Flock Hill area have played a large role in shaping the current vegetation present in the area. Historic fires had taken the area from beech forest to low tussock grassland with very little woody vegetation present by the start of the 20th century (Young *et al*. 2016). Since the 1950s there has been a reduction in the frequency of fires in the Flock Hill area have decreased due to a reduction in livestock grazing (Ledgard 2001, Young *et al*. 2016) Young *et al*. (2016) looked at the vegetation change at Cass, which is located a small distance to the north of Flock Hill Station, over the last one hundred years. They found that since the fire regime had lessened that the native woody vegetation had expanded. It is

likely that overtime this native shrubland will revert back to beech forest at Cass and this is also likely to be the case throughout the eastern South Island (Young *et al*. 2016). While it is possible that fire can help reduce wilding pine numbers in New Zealand, fire can be a double edged sword as it is likely that the predicted increase in the fire regime in the Canterbury high country under future climate change will have a large negative impact on the recovering native vegetation and will likely prevent the succession back to beech forest.

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References

Alexander, M.A (1980) Four fire scar records on lodgepole Pine (*Pinus contorta* Dougl.) in North-Central Colorado. The Southwestern Naturalist 25(3): 432-434.

Allen, R.B (1988) A forest succession in the Catlins Ecological Region, South-East Otago, New Zealand. New Zealand Journal of Ecology 11: 21-29.

Bellingham (1998) Shrub succession and invisibility in a New Zealand montane grassland. Australian Journal of Ecology 23: 562-573.

Bellingham, P.J, Coomes, D.A (2003) Grazing and community structure as determinants of invasion success by Scotch broom in a New Zealand montane shrubland. Diversity and Distributions 9(1): 19-28.

Berglund, B.E, Kitagawa, J, Lageras, P, Nakamura, K, Sasaki, N, Yasuda, Y (2014) Traditional farming landscapes for sustainable living in Scandinavia and Japan: Global revival through the Satoyama initiative. AMBIO 43(5): 559-578.

Brown, K.J, Giesecke, T (2014) Holocene fire disturbance in the boreal forest of central Sweden. Boreas 43(3): 639-651.

Bond, W.J, Dickinson, K.J.M, Marks, A.F (2004) What limits the spread of fire-dependent vegetation? Evidence from geographic variation of serotiny in a New Zealand shrub. Global Ecology and Biogeography 13(2): 115-127.

Boulant, N, Kunstler, G, Rambal, S, Lepart, J (2008) Seed supply, drought, and grazing determine spatio-temporal patterns of recruitment for native and introduced invasive pines in grasslands. Diversity and Distributions 14(5): 862-874.

Clear, J.L, Seppa, H, Kousmanen, Niina, Bradshaw, R.H.W (2015) Holocene stand-scale vegetation dynamics and fire history of an old growth spruce forest in southern Finland. Vegetation History and Archaeobotany 24(6): 731-741.

Cobar-carranza, A.J, Garcia, R.A, Pauchard, A, Pena, E (2014) Effects of *Pinus contorta* invasion on forest fuel properties and its potential implications on the fire regime of *Araucaria araucana* and *Nothofagus antarctica* forests. Biological Invasions 16(11): 2273-2291.

Coops, N.C, Waring, R.H (2011) A process-based approach to estimate lodgepole pine (*Pinus contorta* Dougl.) distribution in the Pacific Northwest under climate change. Climate Change 105(1): 313-328.

Cox, S.C, Barrell, D.J.A (2007) Geology of the Aoraki area.Institute of Geological and Nuclear Sciences. 1:250000 geological map 15. 1 sheet + 71p. Lower Hutt, New Zealand. GNS Science.

Dahlstrom, A, Cousins, S.A.O, Eriksson, O (2006) The history (1620-2003) of land use, people and livestock, and the relationship to present plant species diversity in a rural landscape in Sweden. Environment and History 12(2): 191-212.

Davis, M, Coker, G, Howell, C, Henley, D (2011) Establishment of *Pseudotsuga menziesii* and *Pinus nigra* seedlings in *Kunzea ericoides* and *Leptospermum scoparium* shrubland communities. New Zealand Journal of Ecology 35(3): 280-286.

Drobyshev, I, Niklasson, M, Linderholm, H.W (2012) Forest fire activity in Sweden: Climate controls and geographic patterns in 20th century.Agricultural and Forest Meteorology 154: 174-186.

Edwards, M, Franklin-Smith, L, Clarke, C, Baker, J, Hill, S, Gallagher, K (2015) The role of fire in the mid-Holocene arrival and expansion of lodgepole pine (*Pinus contorta* var. *Latifolia* Engelm. ex S.Watson) in Yukon, Canada. The Holocene 25(1): 64-78.

Engelmark, O, Bradshaw, R, Bergeron, Y (1993) Disturbance dynamics in boreal forest: Introduction. Journal of Vegetation Science 4(6): 730-732.

Engelmark, O, Kullman, L, Bergeron, Y (1994) Fire and age structure of Scots pine and Norway spruce in Northern Sweden during the past 700 years. New Phytology 126(1): 163-168.

Engelmark, O, Hofgaard, A, Arnborg, T (1998) Successional trends 219 years after fire in an old *Pinus sylvestris* stand in Northern Sweden. Journal of Vegetation Science 9(4): 583-592.

Engelmark, O, Sjoberg, K, Andersson, B, Rosvall, O, Agren, G.I, Baker, W.L, Barklund, P, Bjorkman, C, Despain, D.G, Elfving, B, Ennos, R.A, Karlman, M, Knecht, M.F, Knight, D.H, Ledgard, N.J, Lindelow, A, Nilsson, C, Peterken, G.F, Sorlin, S, Sykes, M.T (2001) Ecological effects and management aspects of an exotic tree species: the case of lodgepole pine in Sweden. Forest Ecology and Management 141(2): 3-13.

Eriksson, A-M, Olsson, J, Jonsson, B.G, Toivanen, S, Edman, M (2013) Effects of restoration fire on dead wood heterogeneity and availability in three *Pinus sylvestris* forests in Sweden. Silva Fennica 47(2) article id 954. 15p.

Eriksson, O, Cousins, S.A.O, Bruun, H.H (2002) Land-use history and fragmentation of traditionally managed grasslands in Scandinavia. Journal of Vegetation Science 13(5): 743-748.

Esteban, L.G, De Palacios, P, Rodriguez-Losada Aguado, L (2010) *Abies pinsapo* forest in Spain and Marocco: threats and conservation. Oryx 44(2): 276-284.

Gous, S.F, Watts, M.S, Richardson, B, Kimberly, M.O (2010) Herbicide screening pot trail for wilding conifer control (*Pinus contorta, P. mugo* and *Pseudotsuga menziesii*). New Zealand Journal of Forestry 55(1): 153-159.

Gous, S, Raal, P, Watt, M.S (2014) Dense wilding conifer control with aerially applied herbicides in New Zealand. New Zealand Journal of Forestry Science 44(4): 1-5.

Gous, S, Raal, P, Watts, M.S (2015) The evaluation of aerially applied triclopyr mixtures for the control of dense infestations of wilding *Pinus contorta* in New Zealand. New Zealand Journal of Forestry Sciences 45(1): 1-4

Griesman, A, Gaillard, M-J (2009) The role of climate variability and fire in early and mid Holocene forest dynamics of Southern Sweden. Journal of Quaternary Science 24(6): 593-611.

Groven, R, Niklasson, M (2005) Anthropogenic impact on past and present fire regimes in a boreal forest landscape of southeastern Norway. Canadian Journal of Forest Research 35(11): 2719-2726.

Guild, D, Dudfield, M (2010) A history of fire in the forest and rural landscape in New Zealand – Part 2, post 1830 influences, and implications for future fire management. New Zealand Journal of Forestry 54(4): 31-38

Hagner, S (1983) *Pinus contorta*: Sweden's third conifer. Forest Ecology and Management 6(3): 185-199.

Harvey, B.J, Donato, D.C, Turner, M.G (2016) High and dry: post-fire tree seedling establishment in subalpine forests decreases with post-fire drought and large stand-replacing burn patches. Global Ecology and Biogeography 25(6): 655-669.

Hejcman, M, Hejcmanova, Pavlu, V, Benes, J (2013) Origin and history of grasslands in Central Europe – a review. Grass and Forage Science 68(3): 345-363.

Hellberg, E, Niklasson, M, Granstrom, A (2004) Influence of landscape structure on patterns of forest fires in boreal forest landscapes in Sweden. Canadian Journal of Forest Research 34(2): 332-338.

Humphrey, J.W, Patterson, G.S (2000) Effects of late summer cattle grazing on the diversity of riparian pasture vegetation in an upland conifer forest. Journal of Applied Ecology 37(6): 986-996.

Ivanova, G.A, Ivanov, V.A, Kukavskaya, E.A, Soja, A.J (2010) The frequency of forest fires in Scots pine stands of Tuva, Russia. Environmental Research Letters 5(1):doi:10.1088/1748-9326/5/1/015002

Jakes, P.J, Kelly, L, Langer, E.R (2010) An exploration of a fire-affected community undergoing change in New Zealand. The Australian Journal of Emergency Management 25(3): 48-53.

Johnson, P.N (2001) Vegetation recovery after fire on a southern New Zealand peatland. New Zealand Journal of Botany 39(2): 251-267.

Kashian, D.M, Romme, W.H, Tinker, D.B, Turner, M.G, Ryan, M.G (2013) Postfire changes in forest carbon storage over a 300-year chronosequence of *Pinus contorta*-dominated forest. Ecological Monographs 83(1): 49-66.

Kingery, J.L, Graham, R.T (1991) The effects of cattle grazing on ponderosa pine regeneration. Forestry Chronical 67(3): 245-248.

Knight, D.H, Baker, W.L, Engelmark, O, Nilsson, C (2001) A landscape perspective on the establishment of exotic tree plantations: lodgepole pine (*Pinus contorta*) in Sweden. Forest Ecology and Management 141(1): 131-142.

Kouba, Y, Martinez-garcia, F, de Frutos, A, Alados, C.L (2015) Effects of previous land-use on plant species composition and diversity in Mediterranean forests. PLoS ONE 10(9): e0139031. Doi:10.1371/journal.pone.0139031.

Kuosmanen, N, Fang,K, Bradshaw, R.H.W, Clear, J,L, Seppa, Heilkki (2014) Role of forest fires in Holocene stand-scale dynamics in the unmanaged taiga forest of northwestern Russia. The Holocene 24(11): 1503-1514.

Ledgard, N.J, Belton, M.C (1985) Exotic trees in the Canterbury high country. New Zealand Journal of Forestry Science 15(3): 298-323.

Ledgard (1989) The spread of Douglas Fir into mountain beech forest on Burnt Face, Craigieburn Forest Park.

Ledgard, N.J, Langer, E.R (1999) Wilding prevention: guidelines for minimising the risk of unwanted wilding spread from new plantings of introduced conifers. New Zealand Forest Research Institute Limited.

Ledgard, N (2001) The spread of lodgepole pine (*Pinus contorta,* Dougl.) in New Zealand. Forest Ecology and Management 141(1): 43-57.

Ledgard, N (2002) The spread of Douglas-fir into native forests. New Zealand Journal of Forestry 47(2): 36-38.

Ledgard, N.J (2004) Wilding conifers – New Zealand history and research background. Pages 1-26 in Hill, R.L, Zydenbos, S.M, Bezar, C.M eds, Managing wilding conifers in New Zealand: present and future. New Zealand Plant Protection Society Inc.

Ledgard, N, Knowles, L, De La Mare, P (2005) Douglas-fir – the current New Zealand scene. New Zealand Journal of Forestry 50(1): 13-16.

Ledgard, N.J (2006) Determining the effect of increasing vegetation competition through fertiliser use on the establishment of wildings in unimproved high country grassland. New Zealand Journal of Forestry 51(3): 29-34.

Ledgard, N.J (2008) Assessing risk of the natural regeneration of introduced conifers, or wilding spread. New Zealand Plant Protection 61: 91-97.

Ledgard, N.J, Norton, D.A (2008) The impact of browsing on wilding conifers in the South Island high country. New Zealand Journal of Forestry 52(4): 29-34.

Ledgard, N.J, Paul, T.S.H (2008) Vegetation succession over 30 years of high country grassland invasion by *Pinus contorta*. New Zealand Plant Protection 61: 98-104.

Ledgard, N.J (2009) Wilding control guidelines for farmers and land managers. New Zealand Plant Protection 62: 380-386.

Lehtonen, H, Huttunen, P (1997) History of forest fires in eastern Finland from the fifteenth centurt AD – the possible effects of slash-and-burn cultivation. The Holocene 7(2): 223-228.

Lehtonen, H, Kolstrom, T (2000) Forest fire history in Viena Karelia, Russia. Scandinavian Journal of Forest Research 15(6): 585-590.

Lindbladh, M, Niklasson, M, Nilsson, S.G (2003) Long-time record of fire and open canopy in a high biodiversity forest in southeast Sweden. Biological Conservation 114(2): 231-243.

Maclaren, P (2005) Realistic alternatives to radiata pine in New Zealand – a critical review. New Zealand Journal of Forestry 50(1): 3-10.

McCracken, L.J (1980) Mountain climate in the Craigieburn range, New Zealand. Mountain environments and subalpine tree growth: 41-60.

McIntosh, A.C.S, Macdonald, S.E, Gundale, M.J (2012) Tree species versus regional controls on ecosystem properties and processes: an example using introduced *Pinus contorta* in Swedish boreal forests. Canadian Journal of Forestry Research 42(7): 1228-1238.

Miller & Ecroyd (1987) FRI Bulletin No 124 Introduced forest trees in NZ: recognition, role and seed source 2. Pinus contorta.MoF, FRI P Bag, Rotorua. 12pp.

Muir, P.S, Lotan, J.E (1985) Disturbance history and serotiny of *Pinus contorta* in western Montana. Ecology 66(5): 1658-1668.

Norman, S.P, Taylor, A.H (2005) Pine forest expansion along a forest-meadow ecotone in northeastern California, USA. Forest Ecology and Management 215(1): 51-68.

Ogden, J, Deng, Y, Boswijk, G, Sandiford, A (2003) Vegetation changes since early Maori fires in Waipoua Forest Northern New Zealand. Journal of Archaeological Science 30(6): 753-767.

Ohlson, M, Brown, K.J, Birks, H.J.B, Grytnes, J-A, Hornberg, G, Niklasson, M, Seppa, H, Bradshaw, R.H.W (2011) Invasion of Norway spruce diversifies the fire regime in boreal European forests. Journal of Ecology 99(2): 395-403.

Paul, T.H.S, Ledgard, N.J (2008) Effect of felled wilding pines on plant growth in high country grasslands. New Zealand Plant Protection 61: 105-110.

Paul, T.H.S, Ledgard, N.J (2009) Vegetation succession associated with wilding conifer removal. New Zealand Plant Protection 62: 374-379.

Perry, G.L.W, Wilmshurst, J.M, McGlone, M.S (2014) Ecology and long-term history of fire in New Zealand. New Zealand Journal of Ecology 38(2): 157-176.

Petrie, M.D, Wildeman, A.M, Bradford, J.B, Hubbard, R.M, Lauenroth, W.K (2016) A review of precipitation and temperature control on seedling emergence and establishment for ponderosa and lodgepole pine forest regeneration. Forest Ecology and Management 361: 328-338.

Pierce, A.D, Taylor, A.H (2011) Fire severity and seed source influence lodgepole pine (*Pinus contorta* var. *murrayana*) regeneration in the southern cascades, Lassen volcanic National Park, California. Landscape Ecology 26(2): 225-237.

Pitkanen, A, Huttunen, P, Jungner, H, Tolonen, K (2002) A 10 000 year local forest fire history in a dry heath forest site in eastern Finland, reconstructed from charcoal layer records of a small mire. Canadian Journal of Forest Research 32(10): 1875-1880.

Pole, M, Vajda, V (2009) A new terrestrial Cretaceous-Paleogene site in New Zealand – turnover in macroflora confirmed by palynology. Cretaceous Research 30(4): 917-938.

Poppay, I, Field, R (1996) Grazing animals as weed control agents. Weed Technology 10(1): 217-231.

Primack, R.B (1978) Effects of grazing on indigenous shrubs in tussock grassland at Cass, Canterbury, New Zealand. New Zealand Journal of Botany 16(4): 461-469.

Relph, D.H (1957) The vegetation of the Castle Hill basin. New Zealand Geographer 13(1): 41-55.

Sage, D.J.M, Norton, D.A, Espie, P.R (2009) Effects of grazing exclusion on the woody weed *Rosa rubiginosa* in high country short tussock grassland. New Zealand Journal of Agricultural Research 52(2): 123-128.

Schworer, C, Colombaroli, D, Kaltenrieder, P, Rey, F, Tinner, W (2015) Early human impact (5000-3000 BC) affects mountain forest dynamics in the Alps. Journal of Ecology 103(2): 281-295.

Shanks, A, Glenny, D, Gibson, R, Rosser, K, Roozen, D, Phillipson, S, Steven, J, Arand, J (1990) Coleridge, Craigieburn and Cass ecological districts. P 306. Wellington, New Zealand. Department Of Conservaion.

Smithwick, E.A.H, Turner, M,G, Metzger, K.L, Balser, T.C (2005) Variation in NH_4^+ mineralisation and microbial communities with stand age in lodgepole pine (*Pinus contorta*) forests, Yellowstone National Park (USA). Soil Biology & Biochemistry 37: 1546- 1559.

Smithwick, E.A.H, Ryan, M.G, Kashian, D.M, Romme, W.H, Tinker, D.B, Turner, M.G (2009) Modeling the effects of fire and climate change on carbon and nitrogen storage in lodgepole pine (*Pinus contorta*) stands. Global Change Biology 15(3): 535-548.

Strong, W.L, Hills, L.V (2013) Holocene migration of lodgepole pine (*Pinus contorta* var *latifolia*) in southern Yukon, Canada. The Holocene 23(9): 1340-1349.

Tasser, E, Tappeiner, U (2002) Impact of land use changes on mountain vegetation. Applied Vegetation Science 5(2): 173-184.

Tryterud, E (2003) Forest fire history in Norway: from fire-disturbed pine forests to fire-free spruce forests. Ecography 26(2): 161-170.

Turner, M.G, Turner, D.M, Romme, W.H, Tinker, D.B (2007) Cone production in young post-fire *Pinus contorta* stands in Greater Yellowstone (USA). Forest Ecology and Management 242(2): 119-126.

Vanha-Majamaa, I, Lilja, S, Ryoma, R, Kotiaho, J.S, laaka-lindberg, S, Lindberg, H, Puttonen, P, Tamminen, P, Toivanen, T, kuuluvainen, T (2007) Rehabilitating boreal forest structure and species composition in Finland through logging, dead wood creation and fire: The EVO experiment. Forest Ecology and Management 250(1): 77-88.

Vittoz, P, Rulence, B, Largey, T, Frelechoux, F (2008) Effects of climate and land-use change on the establishment and growth of cembran pine (*Pinus cembra* L.) over the altitudinal treeline ecotone in the central Swiss Alps. Arctic, Antarctic, and Alpine Research 40(1): 225-232.

Wardle, P (1985a) New Zealand timberlines. 1. Growth and survival of native and introduced tree species in the Craigieburn Range, Canterbury. New Zealand Journal of Botany 23(2): 219-234.

Wardle, P (1985b) New Zealand timberlines. 3. A synthesis. New Zealand Journal of Botany 23(2): 263-271.

Wardle, P (2001) Holocene forest fires in the upper Clutha district, Otago, New Zealand. New Zealand Journal of Botany 39(3): 523-542.

Williams, P.A, Kean, J.M, Buxton, R.P (2010) Multiple factors determine the rate of increase of an invading non-native tree in New Zealand. Biological Invasions 12(5): 1377-1388.

Wiser, S.K, Allen, R.B, Platt, K.H (1997) Mountain beech forest succession after a fire at Mount Thomas Forest, Canterbury, New Zealand. New Zealand Journal of Botany 35(4): 505-515.

Wooster, M.J, Zhang, Y.H (2004) Boreal forest fires burn less intensely in Russia than in North America. Geophysical Research Letters 31(20): L20505, doi:10.1029/2004GLO20805.

Young, L.M, Norton, D.A, Lambert, M.T (2016) One hundred years of vegetation change at Cass, eastern South Island high country. New Zealand Journal of Ecology 40(3): 289-301